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Sources, Transfer, and Transformation of Nr in Environmental Systems

The Committee was charged with identifying and analyzing, from a scientific perspective, the problems Nr presents in the environment and the links among them. This chapter addresses two aspects of the Committee's work. The first aspect is the introduction of Nr into U.S. environmental systems from fossil fuel combustion and from food production, and the second aspect is the fate of Nr after it is emitted to the atmosphere by fossil fuel combustion or lost to the air, water and soils from agricultural production systems. The Nr budgets and calculations for the U.S. exclude Alaska and Hawaii because data were not available for these areas. Most of the Nr introduced into the environment comes from the Haber-Boesch process. Haber-Boesch contributes three to four times the Nr introduced from fossil fuel combustion and most of the Haber-Boesch introduction of Nr comes through agriculture.

2.1. Nr Flux in the Environment

As stated previously, although N is a major required nutrient that governs growth and reproduction of living organisms, Nr losses to the environment from human sources have a profound effect on air, water and soil quality. Human consumption of energy to sustain economic development results in emissions of NO_x to the atmosphere via fossil fuel combustion. Consumption of food to meet nutritional requirements of a growing population results in agricultural emissions of NH_3 , urban and industrial emissions of NO_x and N_2O , and losses of NO_3^- and other N compounds to water bodies due to

leaching and runoff. Once released into the atmosphere by either human or natural processes, these Nr compounds undergo transformation through atmospheric reactions (e.g., gas-to-particle conversion), transport associated with wind, and finally wet and dry deposition. Reactive nitrogen lost from agricultural and peopled systems can enter groundwater, streams, lakes, estuaries, and coastal waters where the Nr can also undergo transformation mediated by a wide range of biotic and abiotic processes. The introduction of Nr into agroecosystems provides much of the world's food. The losses of Nr to the environment throughout the food production process and during fossil fuel combustion contribute to many of the major environmental problems of today. The impacts of Nr on humans and ecosystems are discussed in the Executive Summary and Chapter 3 of this report.

Some key issues concerning management of Nr in the U.S. environment are the introduction of new Nr by imports, fertilizer production, cultivation-induced biological nitrogen fixation (C-BNF), and fossil fuel combustion. Other important issues are the distribution of Nr within agricultural systems and populated systems and redistribution of Nr through losses from those systems to the environment (Figure 1). National-level values for Nr fluxes are displayed in Table 1. Fluxes that represent the introduction of new Nr into the U.S. are marked with an asterisk and illustrated in Figure 1. In specific sections of this report these values have been used to more clearly determine the flux and fate of Nr in the U.S. Figure 2 illustrates sources of Nr introduced into the United States.

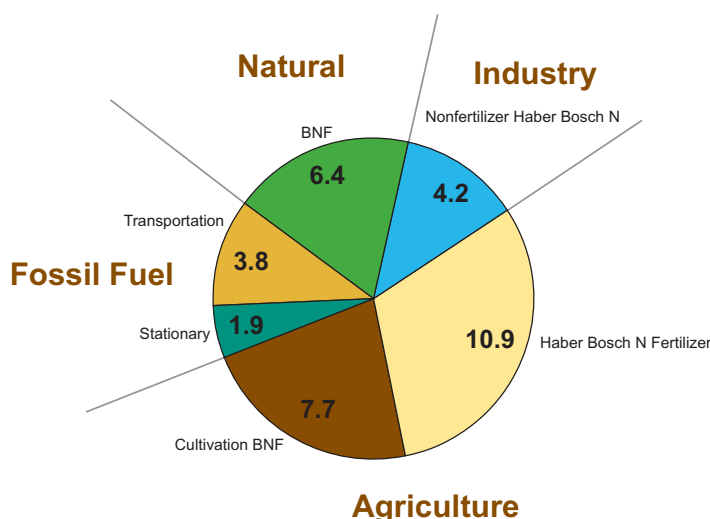


Figure 2: Sources of reactive nitrogen (Nr) introduced into the United States in 2002 (Tg N/yr).

Table 1: Nr fluxes for the United States, Tg N in 2002.^a

Nr inputs to the <i>Atmospheric</i> environmental system	Tg N/yr*	%
N₂O-N emissions¹	0.8	8
Agriculture - livestock (manure) N ₂ O-N	0.03	
Agriculture – soil management N ₂ O-N	0.5	
Agriculture - field burning agricultural residues	0.001	
Fossil fuel combustion - transportation*	0.1	
Miscellaneous	0.1	
NH_x-N emissions²	3.1	31
Agriculture: livestock NH ₃ -N	1.6	
Agriculture: fertilizer NH ₃ -N	0.9	
Agriculture: other NH ₃ -N	0.1	
Fossil fuel combustion – transportation *	0.2	
Fossil fuel combustion - utility & industry *	0.03	
Other combustion	0.2	
Miscellaneous	0.1	
NO_x-N emissions²	6.2	61
Biogenic from soils	0.3	
Fossil fuel combustion – transportation *	3.5	
Fossil fuel combustion - utility & industry *	1.9	
Other combustion	0.4	
Miscellaneous	0.2	
Total <i>Atmospheric</i> inputs	10.0	100
Nr inputs to the <i>Terrestrial</i> environmental system		
Atmospheric N deposition^b	6.9	16
Organic N ³	2.1	
Inorganic NO _y -N ⁴	2.7	
Inorganic-NH _x -N ⁴	2.1	
N fixation in cultivated croplands *⁵	7.7	18
Soybeans*	3.3	
Alfalfa*	2.1	
Other leguminous hay *	1.8	
Pasture*	0.5	
Dry beans, peas, lentils *	0.1	
N fixation in non-cultivated vegetation *⁶	6.4	15
N import in commodities *⁷	0.2	0.3
Synthetic N *⁸	15.1	35
Fertilizer use on farms & non-farms	10.9	
Non-fertilizer uses	4.2	
Manure N production⁹	6.0	14
Human waste N¹⁰	1.3	3
Total <i>Terrestrial</i> inputs	43.5	100
Nr inputs to the <i>Aquatic</i> environmental system		
Surface water N flux¹¹	4.8	

*Terms with an asterisk indicate Nr that is created, highlighting where reactive nitrogen is introduced to the environment.

Table 1 Notes

a. The Nr estimates in this table are shown with two significant digits or 0.1 million metric tons N per year (or Tg N/yr) to reflect their uncertainty; occasionally this report will show data to more significant digits, strictly for numerical accuracy. Because numbers are rounded, the sums of source category inputs do not always equal the total reported inputs. Obtaining quantitative estimates of each of the Nr terms and the associated uncertainties remains a major scientific challenge.

b. Reducing the uncertainty in total deposition of atmospheric Nr to the surface of the 48 contiguous United States remains a scientific and policy priority. Based on observations and models, we estimate 5.9 (range 4 - 9) Tg N/yr total anthropogenic Nr deposition to the entire 48 States. The EPA sponsored Community Multiscale Air Quality (CMAQ) Model run yielded a value of 4.8 Tg N/yr. The value shown for the total (6.9 Tg N/yr) reflects the assumption that organo-nitrogen species should be added to the model estimate as 30% of the total.

c. The synthetic N total includes 5.8 Tg N of fertilizer net imports to the United States (8.25 Tg N imported - 2.41 Tg N exported) plus 9.4 Tg N of fertilizer produced in the United States in 2002.

Table 1 Data Sources (all data reflect N fluxes in the United States in 2002)

¹ Emissions, N₂O-N (U.S. EPA Inventory of U.S. Greenhouse Gas Emissions and Sinks, 1990-2006)

² Emissions, NH_x-N; Emissions, NO_x-N (U.S. EPA National Emissions Inventory, release version October 2007)

³ Atmospheric deposition, organic N (30% of total atmospheric N deposition, Neff et al. 2002)

⁴ Atmospheric deposition, inorganic NO_y-N and NH_x-N (U.S. EPA CMAQ model)

⁵ N₂ fixation in cultivated croplands (USDA census of agriculture 2002, literature coefficients)

⁶ N₂ fixation in non-cultivated vegetation (unpublished data estimate after Cleveland and Asner, 1990)

⁷ Net N imports in commodities and fertilizer trade (FAO - FAOSTAT)

⁸ Synthetic N fertilizer use (FAO - FAOSTAT and Association of American Plant Food Control Officials - AAPFCO)

⁹ Manure N production (USDA census of agriculture, literature coefficients)

¹⁰ Human waste N (U.S. Census Bureau population census, literature coefficients)

¹¹ Surface water N flux (USGS SPARROW model, after Alexander et al., 2008)

2.2. Sources of New Nr to the Environment

Creation of “new” Nr in the environment refers to Nr that is either newly fixed within or transported into the United States. This “new” Nr highlights where Nr is introduced into ecosystems. New Nr arises from fossil fuel combustion, food production, and materials production (Table 1).

Fossil fuel combustion emits Nr (mostly NO_x) to the atmosphere. Fossil fuel combustion introduces 3.5 Tg N/yr and 1.9 Tg N/yr of NO_x-N to the atmosphere from transportation, and utility/other industry sources, respectively (Table 1). Another 0.2 Tg N/yr of NH₃-N and 0.1 Tg N/yr of N₂O-N is emitted from the same sources (Table 1). Thus the total amount of Nr created by fossil fuel combustion is 5.7 Tg N/yr, of which > 90% is in the form of NO_x-N.

Synthetic Nr fertilizers are typically produced by the Haber-Bosch process and used primarily in agriculture to support food production. Production of fertilizers within the U.S. introduces Nr into U.S. terrestrial landscapes at the rate of 9.4 Tg N/yr, and net imports of fertilizer via world trade introduce 5.8 Tg N/yr. Of this total (15.2 Tg N/yr), 9.8 Tg N/yr is used as fertilizer on farms and 1.1 Tg N/yr is used on non-farms (i.e., residential and recreational turf-grass and gardens, and in explosives used by the mining industry), and 4.2 Tg N/yr is introduced for non-fertilizer uses, such as for production of plastics, fibers, resins, and for additives to animal feed (Table 1).

Additional Nr is introduced into the U.S. from C-BNF by agricultural legume crops such as soybean and alfalfa (7.7 Tg N/yr), and from imports of N contained in grain and meat (0.15 Tg N/yr) (Table 1).

Thus in 2002, anthropogenic activities introduced a total of 29 Tg N into the U.S., mostly in support of food production, although turf production, industrial uses and fossil fuel combustion were also important sources. Natural sources of Nr in the U.S. are biological nitrogen fixation (BNF) in unmanaged landscapes, and lightning. The former contributes 6.4 Tg N/yr (Table 1) and the latter 0.1 Tg N/yr. Clearly, anthropogenic activities dominate the introduction of Nr into the U.S.

Losses of Nr to the environment in the U.S. occur during fossil fuel combustion and food production. The former occurs immediately, as Nr formation during combustion is inadvertent and the Nr, primarily as NO_x, is emitted directly into the atmosphere. The latter occurs through all stages of food production and consumption. The following four Sections (2.2.1-2.2.4) of this report document the magnitude of the losses of Nr to the environment from the various components of both energy and food production.

2.2.1. *Nr Formation and Losses to the Environment from Fossil Fuel Combustion*

Fossil fuels such as coal, petroleum, and natural gas provide about 80% of all energy production in the U.S. (based on year 2000). When these fuels are burned at high temperatures, NO_x is formed. The source of N is either the N contained in the fossil fuel or the N_2 that makes up about 80% of atmosphere. Fuel-derived N is important in the case of burning coal (which contains N), while atmospheric-derived N_2 is transformed to NO_x during higher temperature processes that occur when gasoline or diesel fuel is burned in motor vehicles (Table 1). As Figure 3 indicates, in the U.S., highway motor vehicles account for the largest anthropogenic source of NO_x (36%), followed by off-highway vehicles, electric utilities, and industrial processes. Emissions from aircraft make up only about 1% of the U.S. total for NO_x , but a large proportion of this is released in free troposphere where lifetimes are long and adverse impacts wide-ranging. As such, continued reductions are encouraged (e.g., EPA Regulatory Announcement: New Emission Standards for New Commercial Aircraft Engines, www.epa.gov/oms/regs/nonroad/aviation/420f05015.htm).

Figure 3 also illustrates that the amount of NO_x (reported as metric tons of N) released from various fossil fuel sources has decreased dramatically from 1970. Total emissions were on the order of 7,400 metric tons in 1970 and decreased to 5,900 in 2002, with further decreases in 2006 to 5,030 metric tons. Overall this represents a decrease of over 30%. The top sources (highway vehicles, off-highway vehicles,

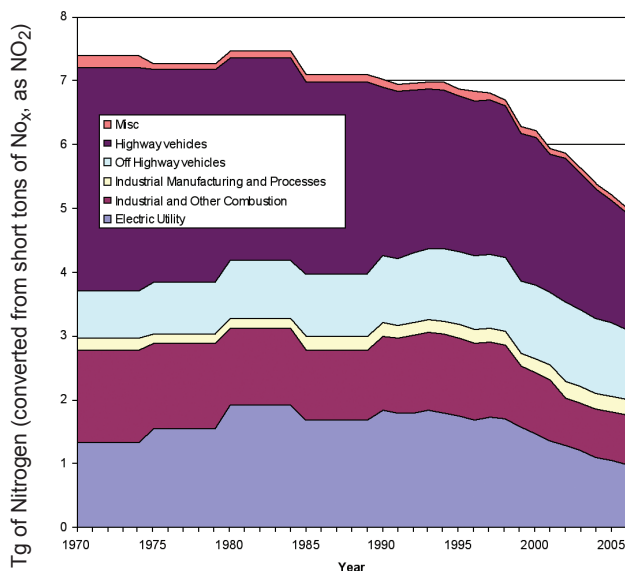


Figure 3: U.S. NO_x emission trends, 1970-2006

Data are reported as thousands of metric tons of N converted from NO_x as NO_2

Data source: www.epa.gov/ttn/chief/trends. More recent information provided by EPA indicates that electric power NO_x emissions decreased 70% between 1990 and 2009 (U.S. EPA, 2010b).

electric utilities, and other industrial and combustion systems) show decreases of 15-30% from 1990 to 2002 (Figure 4). Reductions were the highest for “other” systems followed by electric utilities. These decreases are most likely the result of changes in regulations and control technologies for these stationary systems. More recent preliminary information provided by EPA indicates that electric power NO_x emissions may have decreased 70% between 1990 and 2009 and that the electric power sector’s NO_x emissions now account for about 12% of anthropogenic NO_x emissions in the U.S. (U.S. EPA, 2010b). To a lesser extent, changes in highway vehicle regulations and the removal of older fleets from the road has resulted in a decrease of approximately 15%. This decrease however, is accompanied by an increase in miles traveled, which suggests that the actual decrease in a single vehicle is larger. Off-highway vehicles showed an increase in emissions, potentially due to better quantification of these sources. Such sources include locomotives and marine engines. EPA is in the process of implementing a number of regulations that will reduce NO_x emission from mobile sources (see Appendix F). Figure 5 (provided by EPA) projects decreases in U.S. mobile-source NO_x emissions. The implications of these recent regulations are not reflected in the quantitative analyses presented in this report. However, additional control of these and other sources could further decrease emissions. In fact, technological development in the locomotive industry shows that decreases of approximately 70% are possible. Further decreases would require more innovative, expensive methods such as selective catalytic reduction

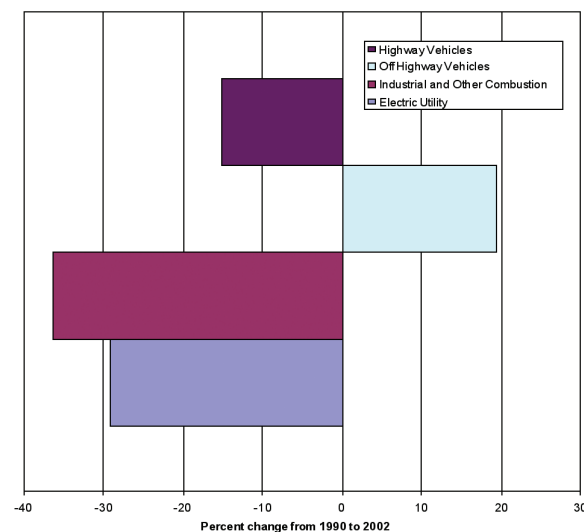


Figure 4: Percent reductions in NO_x emissions, 1990-2002, from different sources (off-road, on-road, power generation, etc.)

Data source: www.epa.gov/ttn/chief/trends. More recent preliminary information (U.S. Environmental Protection Agency, 2010b) indicates that electric power NO_x emissions may have decreased 70% between 1990 and 2009.

(SCR) with urea injection. Engine manufacturers are also investigating using SCR systems for diesels. However, it must be noted that these systems emit small amounts of NH_3 and must be operated properly to avoid trading off NO_x emissions for NH_3 emissions.

It should also be noted that it is difficult to control nitrogen emissions by regulating one source solely. As seen in Table 2 (data taken from 2001 for illustrative purposes have been reported with a higher degree of precision than is likely to be known), in Texas, fuel combustion

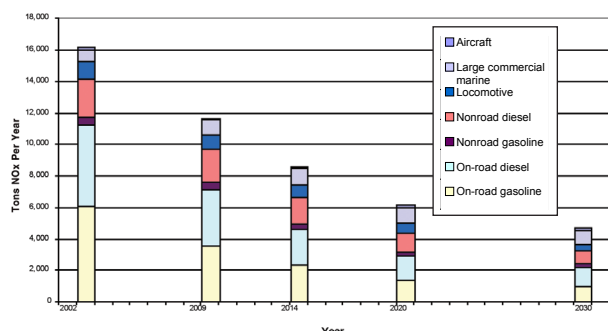


Figure 5: Mobile source NO_x emission inventories

Source: Figure provided by Margaret Zawacki of the U.S. EPA Office of Transportation and Air Quality. Inventory data used to develop this figure are available in EPA's final rule, Control of Emissions from New Marine Compression-Ignition Engines at or Above 30 Liters per Cylinder (www.epa.gov/otaq/regs/nonroad/marine/ci/420r09019.pdf), except for onroad emissions, which EPA generated by running MOVES2010 (www.epa.gov/otaq/models/moves/index.htm) at the national-month level.

sources are on the same order as emissions from highway vehicles. By comparison, in California, highway and off-highway vehicles are the dominant source of nitrogen emissions (over 75%). These results can be attributed to differences in the two states' industries and power plants. In Texas, almost 40% of the power generation is from coal-fired plants. On the other hand, California imports most of its coal-fired power and generates its own power predominantly from other sources, such as natural gas (50%), hydro and nuclear (33%). Table 2 also shows emissions in Florida, Ohio, and Illinois. The emission of NO_x from highway vehicles is likely related to population. For example, the estimated population of California in 2006 was 36.4 million people versus Ohio and Illinois which are on the order of 11-12 million.

2.2.2. Nr Inputs and Losses to the Environment from Crop Agriculture

Agriculture uses more Nr and accounts for more Nr losses to the environment than any other economic sector. Synthetic fertilizers are the largest sources of Nr input to agricultural systems. The next largest source is N fixation in cultivated croplands (Table 1). The major pathways by which Nr is lost from these systems include NO_3^- losses from leaching, runoff and erosion and gaseous emissions via volatilization of NH_3 and NO_x and nitrification/denitrification. Similar loss pathways occur for Nr that cycles through livestock systems, which also account for a large portion of Nr flux (predominantly as NH_3) in animal agricultural systems (Aneja et al., 2006). Therefore, assessment of Nr impacts on the environment and development of strategies to minimize negative impact should be based on a thorough understanding and accurate accounting

Table 2: Examples of multiple sources from states with high NO_x emissions based on 2001 data, and tons of NO_x as NO_2

	TX	CA	FL	OH	IL
Fuel Combustion – Electric Util.	91,441	8,441	87,489	93,792	59,124
Fuel Combustion – Industrial	98,978	31,237	11,792	17,300	26,481
Fuel Combustion – Other	9,222	21,407	5,707	12,974	10,894
Industrial Processes	25,584	13,786	5,933	8,123	7,122
Highway Vehicles	164,937	182,471	116,889	83,593	78,278
Off-Highway Vehicles	106,162	85,064	38,475	46,239	52,797
Miscellaneous Sources	4,807	7,882	13,110	1,526	999
TOTAL, metric tons	501,151	350,301	279,778	263,561	235,817

Source: Data derived from the 2001 information obtained at: www.epa.gov/air/data/.

of Nr fluxes in both crop and livestock systems, and the trends in management practices that have greatest influence on Nr loss to the environment from these systems (Aneja et al, 2008a,c).

As previously noted, in the past 60 years N fertilizers have had a beneficial effect on agriculture both nationally and globally by increasing crop yields. However, the high loading of Nr from agricultural nutrient sources has led to deleterious effects on the environment, such as decreased visibility from increased aerosol production and elevated N concentration in the atmosphere, ground, and surface waters (Galloway et al., 2003).

Nitrogen fertilizer use information

Obtaining accurate data on fertilizer use is a critical first step in understanding Nr cycles in agriculture. There are several sources of data reporting fertilizer usage but it is not clear whether data quality are sufficient for assessing environmental impact. Although the Uniform Fertilizer Tonnage Reporting System (UFTRS) was developed to collect fees to fund the consumer protection mission of state chemists and fertilizer regulatory control officials, it also provides data on fertilizer sales in many states, which in turn are used by many agencies and environmental scientists to estimate consumption and use of nitrogenous fertilizers in the U.S. The Association of American Plant Food Control Officials (AAPFCO) tallies and publishes the statewide fertilizer sales data annually (Terry and Kirby, 2006). Information published by AAPFCO is a widely used source of fertilizer use data. It is typically assumed that fertilizers are used in the same region in which they were sold. The annual data published by AAPFCO, which are based on commercial fertilizer sold (and taxed in some but not all states), is the only state-level data source available. This state-level data source includes fertilizer sales for both agricultural and non-agricultural purposes. These state-level data must then be allocated to counties, regions, or watersheds in the states, and the algorithms used for this process are based on a number of assumptions that address dealer/farmer storage, inventories, and cross-state sales issues (personal communication, Stan Daberkow, USDA-ERS).

The U.S. Department of Agriculture (USDA) National Agricultural Statistics Service (NASS) fertilizer usage data represents another source of information derived from farmer “agricultural chemical use” surveys that provide information in six categories: field crops, fruits and vegetables, nurseries/floriculture, livestock use, and post-harvest application. NASS periodically collects fertilizer, pesticide, and pest management data from a stratified random sample of farmers at the field level. The NASS report represents a useful data source but also would require extrapolation across reported crop acreage to represent a complete sample of application rates.

The UFTRS was not designed to track the source of inorganic nutrients applied to agricultural land on the geographic scale needed for watershed modeling. The system only tracks sales of synthetic fertilizers and not manure or biosolids applied to farmland. In addition, geographical data associated with each sale may or may not be near the actual point of application. However, given either regulatory or legislative changes (data reporting is mandated through each state’s fertilizer law), it could be possible to refine the current system used by each state’s department of agriculture to generate more precise data for improved modeling of watershed-scale nutrient mass balances. Those changes would help target interventions and extension programs to improve nutrient management and reduce nutrient losses. The lack of potential funding and the necessity to coordinate all the states involved limit the practicality of such an approach.

State departments of agriculture have considered ways to improve the reporting system. Such improvements could include:

1. An assessment to determine the needs for fertilizer usage data, the accuracy of the current data collection methods, and whether methods require revision to meet highest priority needs
2. Improvements in the database format and web-based access
3. The identification of funding sources to support development of a more accurate, accessible, and comprehensive database system
4. Education and outreach to improve precision of reported fertilizer tonnage, including a clear distinction between nutrients used in crop, livestock, and non-agricultural operations

In addition, the information could be refined to reflect site-specific data layers, although that would require development of a geospatial framework (and legal authority) to encourage reporting at the retail level where it is possible to collect geographic information.

The Chesapeake Bay watershed provides a good example of the fertilizer data dilemma. While the fertilizer tonnage that is currently being utilized to calibrate the Chesapeake Bay Program Watershed Model is relatively accurate, the county-specific tonnage may have an accuracy of only ± 20 to 50% (Chesapeake Bay Scientific Technical Advisory Committee, 2007). For example, in a recent year, 17% of the reported tonnage was reported without an identified use, and there are indications some tonnage may have been reported more than once through the distribution chain (Chesapeake Bay Scientific Technical Advisory Committee, 2007). It is also possible that fertilizer reported for crop agriculture may actually have been used for lawn and turf, forestry, or other non-agriculture applications.

Nitrogen fertilizer application data collected on a specific crop-by-crop basis that can be associated with

crop yields and location are essential for assessing both use patterns and efficiency. As previously mentioned, the USDA-NASS maintains a database on N fertilizer rates applied to the major crops (corn, wheat, cotton, soybeans, and occasionally other crops). The data are gathered from farmer surveys conducted every other year. These data represent a source of useful information (*Protocols for Farming Reporting*, Mark R. Miller, USDA, NASS).

Data derived from NASS farmer surveys include six categories: field crops, fruits and vegetables, nurseries/floriculture, livestock use, and post-harvest application. For each group, NASS collects fertilizer, pesticide, and pest management data on a stratified random sample of farmers at the field level. One field represents an entire farm for each sample in the field crops survey. Fruit and vegetable information are collected for the entire farm. If the field chosen for sampling has had manure applied in conjunction with inorganic fertilizer, only the inorganic portion will be reported because the survey does not ask about manure.

Certain core crops are surveyed every other year on an even/odd basis and states are selected to cover at least 80% of planted acres. There has been a recent five-year gap in the collection of NASS data for corn and other commodity crops. NASS was scheduled to resume coverage of corn and other commodity crops in 2010. This is a critical data gap and it is a problem given the large changes in corn price and production area during this period. Those data are needed to make progress in assessing fertilizer use and efficiency for major crops in the U.S. USDA NASS must resume their yearly data

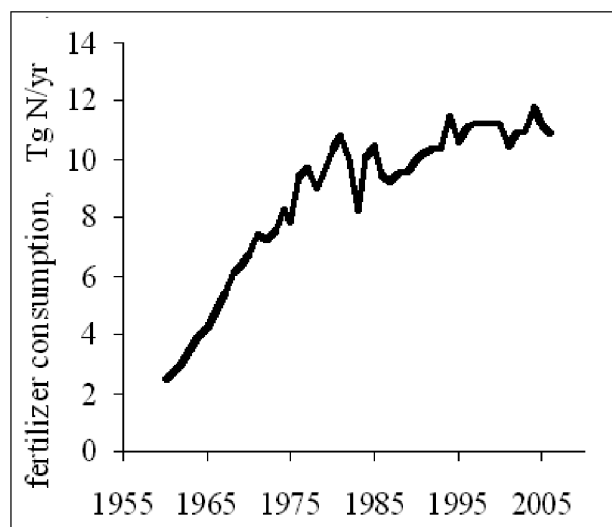


Figure 6: Fertilizer consumption in the United States, 1960 to 2006

Source: Slater et al., 2010. Reprinted with permission from the Association of American Plant Food Control Officials.

collection for commodity crops (Chemical Use Survey). Potential environmental impacts of increased N inputs associated with expanded corn acreage for biofuel production cannot be properly evaluated in the absence of such critical nutrient management data.

Based on the NASS survey data, the USDA Economic Research Service released a report on fertilizer use that provided data on fertilizer consumption and type of fertilizer used from 1960-2006 (Figure 6) and types of fertilizers used (Table 3) (USDA, 2008). The share of crop area receiving fertilizer and fertilizer use per receiving acre, by nutrient, are presented for the major producing states for corn, cotton, soybeans, and wheat. Additional data include fertilizer farm prices and indices of wholesale fertilizer price.

Finding 1: Crop agriculture receives 60% of U.S. annual new Nr inputs from anthropogenic sources (9.8 Tg from N fertilizer, 7.7 Tg from crop BNF versus 29 Tg total) and accounts for 58% (7.6 Tg) of total U.S. Nr losses from terrestrial systems to air and aquatic ecosystems. However, current monitoring of fertilizer use statistics by federal agencies is inadequate to accurately track trends in quantities and fate of N applied to major crops and the geospatial pattern by major watersheds.

Recommendation 1: *The Committee recommends increasing the specificity and regularity of data acquisition for fertilizer application to major agricultural crops in terms of timing and at a sufficiently small application scale (and also for urban residential and recreational turf) by county (or watershed) to better inform decision-making about policies and mitigation options for reducing Nr load in these systems, and to facilitate monitoring and evaluation of impact from implemented policies and mitigation efforts.*

Nitrogen fertilizer use efficiency

Nitrogen fertilizer use efficiency (NFUE) is critical because higher use efficiency leaves less N remaining to create potential environmental problems. Here and throughout this report we define NFUE as the grain yield per unit of applied N, which is the product of two parameters: (1) the proportion of applied N fertilizer that is taken up by the crop, or N fertilizer recovery efficiency [recovery efficiency (RE) in kg N uptake per kg N applied], and (2) the physiological efficiency with which the N taken up by the crop is used to produce economic yield for crops such as grain or fruit [physiological efficiency (PE) in kg yield per kg N uptake] (Cassman et al., 2002)¹³. All else equal, when higher NFUE is achieved without yield reduction, the crop takes up more of the applied N and incorporates it into its biomass, which leaves less of the applied Nr at risk for losses via leaching, volatilization, or denitrification. Fixen (2005) reports that there is substantial opportunity for increasing NFUE through development

¹³ N fertilizer use efficiency (NFUE) is calculated as the ratio of grain yield to the quantity of applied N fertilizer (kg grain/kg applied N).

Table 3: Types and amount of nitrogen fertilizers used in the United States in 2002

Synthetic Nitrogen Fertilizers	Tg N/year	% of total
Other	0.21	2
Urea	2.21	20
N Solutions	2.55	23
Anhydrous NH ₃	2.88	26
Ammonium phosphates and N-P-K blends	2.28	21
Ammonium sulfate, aqua ammonia, ammonium nitrate, and other nitrate and ammonical N fertilizers	0.76	7
Total *	10.89	100

Data from Terry and Kirby, 2006

* Because of number rounding, the sum of individual percentages does not equal 100%.

and adoption of more sophisticated nutrient management decision aids.

In most cropping systems, RE is the most important determinant of NFUE. A recent review of RE for cereals based on field studies around the world, mostly conducted on “small-plot” experiments at research stations, reported mean single year RE values for maize, wheat and rice of 65%, 57%, and 46%, respectively (Ladha et al., 2005). However, crop RE values based on actual measurements in production-scale fields are seldom greater than 50% and often less than 33%. For example, a review of RE in different cropping systems estimated average recoveries of 37% for maize in the north central U.S. (Cassman et al., 2002). It is also important to note that soil N provides the majority of the N taken up by most crops grown on soils with moderate to good soil fertility. For maize in the U.S. Corn Belt, for example, 45-77% of total N uptake was estimated to come from soil N reserves, based on experiments from research stations (Sawyer et al., 2006). Therefore, highest N efficiency and economic return on N inputs are achieved when the amount and timing of applied N is synchronized with the availability of soil N throughout the growing season to minimize both the quantity of N input required and the N losses from soil and applied N sources.

However, there are relatively few data that provide direct measurement of N fertilizer recoveries by major field crops under production-scale conditions. Reducing the uncertainty in estimates of N fertilizer RE is fundamental for prioritization of research and education investments, both in the public and private sectors. While management can substantially improve RE on average, in any given year weather will always be an uncontrolled factor that can significantly influence system efficiency. Weather can influence system efficiency through effects

on crop growth vigor and ability to acquire applied nutrients and through losses of nutrients due to runoff, denitrification, and leaching that can occur in periods of excessive rainfall.

Although total N fertilizer use in the U.S. has increased more slowly in the past two decades (Figure 6), yields of all major crops have continued to increase. Because crop yields are closely related to N uptake (Cassman et al., 2002), these trends imply a steady increase in NFUE and reduced N losses to the environment because more of the applied N is held in crop biomass and harvested grain. Greater NFUE has resulted from two factors. The first factor is a steady improvement in the stress tolerance of corn hybrids (Duvick and Cassman, 1999) that increases crop growth rates and allows sowing at higher plant densities, which together accelerate the establishment of a vigorous root system to intercept and acquire available N in the soil profile. The second factor is the development and adoption of technologies that may improve the congruence between crop N demand and the N supply for indigenous soil resources and applied N. Examples of such technologies include soil testing for residual nitrate and adjusting N fertilizer rates accordingly, split N fertilizer applications, fertigation (the application of nutrients through irrigation systems), site-specific management, and new fertilizer formulations (e.g., controlled release, nitrification inhibitors). For maize, which receives the largest share of total N fertilizer in the U.S. (44% in 2005), NFUE decreased markedly in the 1960s because N fertilizer rates rose more quickly than maize yields. However, with recognition of negative impact from over-application of N and associated N losses to the environment (especially with regard to water quality) investment in research and education to improve N fertilizer efficiency resulted in more than 50% increase in NFUE from 1974-76 to 2002-05 (Figure 7). Similar

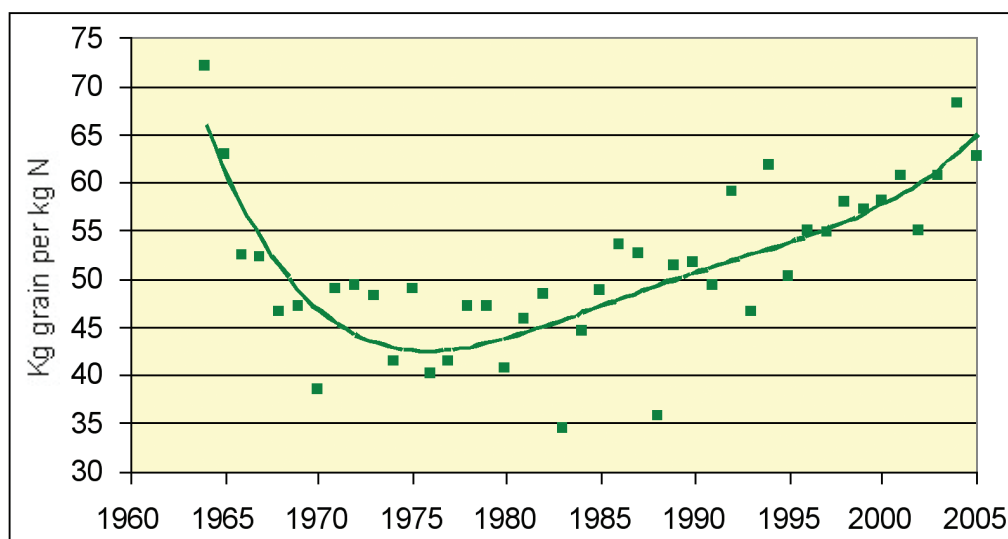


Figure 7: Trends in corn grain produced per unit of applied fertilizer N (NFUE) in the United States

Source: Adapted from Fixen and West, 2002 (Figure 6). Adapted with permission; Copyright 2002, Springer Science+Business Media B.V. on behalf of the Royal Swedish Academy of Sciences.

improvements have been documented for rice production in Japan and for overall crop production in Canada.

Despite these steady improvements, current levels of N fertilizer uptake efficiency appear to be relatively low, although data from production-scale studies are few (Cassman et al., 2002). Most farmers do not use best management practices (BMPs) with regard to nitrogen fertilizer management. For example, a recent U.S. Department of Agriculture Economic Research Service (USDA-ERS) Agricultural Resources and Environmental Indicators (AREI) report indicates that a majority of farmers still apply N in the fall, which gives the lowest fertilizer uptake efficiency and highest N_r losses compared to application in spring or during the crop growth period (USDA, ERS, 2006). This situation suggests substantial potential for improvement in NFUE and an associated reduction in N_r losses from crop agriculture, especially for maize in the warmer portions of the Corn Belt and other southern and southeast areas where maize is grown. One potential development is the use of controlled release fertilizers that release N in congruence with crop demand during the growing season. Although such fertilizers are already in use on high value horticultural crops, they are currently too expensive for lower value commodity grains such as corn, rice, or wheat. Control of N release should result in higher NFUE where there is high risk for N losses in cereal systems that receive the total amount of applied N in one or two large doses. Production-scale field studies are needed to document the benefits of this and other innovative technologies to improve NFUE.

As producers have significantly increased yields in commodity crops over the past 25 years, it is questionable whether university recommendations for nutrient applications are still current. Many university

recommendations are now 20 to 25 years old. As a corollary to this problem, numerous environmental models of nutrient pollution are still utilizing older yield estimates, which often underestimate crop nutrient uptake and overestimate nutrient losses (Burgholzer, 2007).

A systematic effort needs to be made to update data on crop yields used to estimate nutrient losses. The concept of NFUE should be emphasized as a way to address the need to balance economic *and* environmental goals. In fact, the development and adoption of technologies that improve nitrogen fertilizer efficiency can contribute to more profitable cropping systems through a reduction in fertilizer costs. For example, average NFUE in the U.S. required 1.0 kg of applied N to produce 43 kg of grain yield in the 1974-1976 period, whereas that same amount of N produced 65 kg of grain in the 2003-2005 period (data taken from Figure 7). This gain in efficiency means that it is possible to achieve the 2004 U.S. average corn yield of about 150 bushels per acre (9,444 kg/ha) with 144 lbs per acre (161 kg/ha) of applied N fertilizer (based on the most recent NFUE achieved by U.S. corn producers) versus about 200 lbs per acre (224 kg/ha) of N fertilizer at the 1980 efficiency level.

Nitrogen costs have become extremely volatile, mirroring natural gas prices. In late 2008, N fertilizer prices were more than double the 2006-2007 N fertilizer prices. More recently, N fertilizer prices have fallen back to two thirds of the high following the decline of natural gas prices. If corn can be sold for \$4.00 per bushel (25.5 kg) and N costs \$0.40 a pound (0.45 kg), this is a 10 to 1 price ratio – the same as the \$2.00 corn and \$0.20 nitrogen ratio that was typical from 2000 to 2005. There are also other critical factors in a farmer's N application decisions, such as yield at the margin and weather. In the Corn Belt, one or two years in five may provide extremely

favorable weather for corn production. A producer may view applying some extra N, hoping for good weather, as a reasonable economic gamble. If the yield response is more than half a bushel (12.7 kg) of corn per pound (0.45 kg) of N at the margin or if there is more than one extremely good year in five, the farmer benefits.

Realistically, few farmers calculate their marginal returns from additional N in good years versus average, but the high corn-to-fertilizer price ratio encourages some farmers to plan for a good year and consider a larger N application than might otherwise be appropriate for the N utilization in the four years of lower yield. This presents a real dilemma if the policy goal is to reduce N transfers to the environment, especially in the four years of average or lower yields. Meeting this challenge will require approaches such as the development of real-time, in-season, decision-making tools that allow crop producers to use N fertilizer rates for average yields at planting and during early vegetative growth, and a final top-dressing as required to meet any additional N demand above this amount due to favorable climate and soil conditions that support higher than average yields (Cassman, 1999; Cassman et al., 2002). Robust crop simulation models using real-time climate data at a relatively localized geographic scale will be required to develop such tools.

Another option is to develop new, alternative crop production systems that require less N fertilizer. Such systems may employ legume cover crops, more diverse crop rotations, and tighter integration between crop and livestock production to achieve greater reliance on N inputs from legume N fixation and recycling of N in manure and compost. At issue, however, is whether such systems actually reduce Nr losses to the environment because the same loss mechanisms and pathways operate on N from both commercial fertilizer and organic sources. Also at issue is the indirect land use change impact from widespread adoption of these more diverse cropping systems because they have reduced crop yields per unit land area compared to more simplified crop rotations such as corn-soybeans that receive N fertilizer. Lower yields would require more land in production to meet food demand. Therefore, a key issue is whether the tradeoff in reduced N fertilizer inputs to more diverse crop rotations with organic N inputs would actually result in less Nr losses to the environment compared to conventional cropping systems that require less land to produce the same amount of crop output.

Another approach to reduce Nr losses from agriculture would be to shift and/or adjust cropping systems across the landscape. This would involve changes in land use as well as crops. This approach for parametric reductions in nitrogen was analyzed extensively in the Gulf of Mexico Hypoxia Assessment (Doering et al., 1999). As part of a modeling exercise to address opportunities and consequences for reducing Nr, crop rotations as well as tillage practices and fertilizer inputs were adjusted to

meet successive constraints on excessive Nr while also maximizing consumer and producer welfare to the extent possible. The model favored those crops and cropping practices that had lower Nr leakage. Where the model could not find a crop production system at given locations that allowed positive net returns to the land, that land was taken out of production. At a 20 percent Nr reduction scenario, crop acreage was reduced by about 6%. This analysis was based on crop genetics, rotations, and tillage practices as of the 1990s.

Unintended impacts of lower application rates of nitrogen for crop production

Crop production and environmental quality are potentially lost or gained at the expense of each other. Although leakage of N from crop production systems cannot be eliminated, N losses can be minimized substantially. One way to decrease leakage is to apply less N fertilizer to croplands. For example, Hu et al. (2007), using the SWAT model, predict that decreasing N fertilizer application rates by 10 to 50% of those used in the 1990s in the upper Embrarras River watershed in east central Illinois would decrease NO_3^- output to the river by 10 to 43%. This simple “solution” can cause problems for crop production as yields and crop quality (protein content) may decrease, causing economic loss to the farmer, decreased food quality for the consumer and, at a global scale, a reduction in food security.

Cropping systems managed in a consistent manner over time reach a functional equilibrium between nitrogen inputs and outputs. Because crop yields are closely linked to the quantity of N accumulation in above ground biomass at maturity (Cassman et al., 2002), there would be a proportional decrease in crop yields in response to a decrease in the amount of N fertilizer application. The magnitude of this yield reduction would depend on the magnitude of decrease in the rate of applied N and the efficiency of N uptake from the applied N, as well as interrelationships with the availability of other nutrients. Hence, yield reductions can be mitigated, or even eliminated, if methods and fertilizer formulations used in fertilizer-N application increase the efficiency of N uptake to offset the reduction in the amount of applied N. It is also important to note that reduced or insufficient N rates for crop production risk impairment of long-term soil productivity. Jaynes and Karlen (2005) reported that N rates below the agronomic and economic optimum could degrade the soil resource and decrease soil organic matter over time. Thus care must be exercised in any N rate adjustments to protect soil productivity and to support soil resource sustainability.

Another concern associated with decreasing N fertilizer input to cereal crop production is the effect on crop quality. In Denmark, for example, national efforts to decrease Nr losses to the environment resulted in a large reduction in use of N fertilizer in crop production (Figure 8). This situation occurred as part of Denmark’s

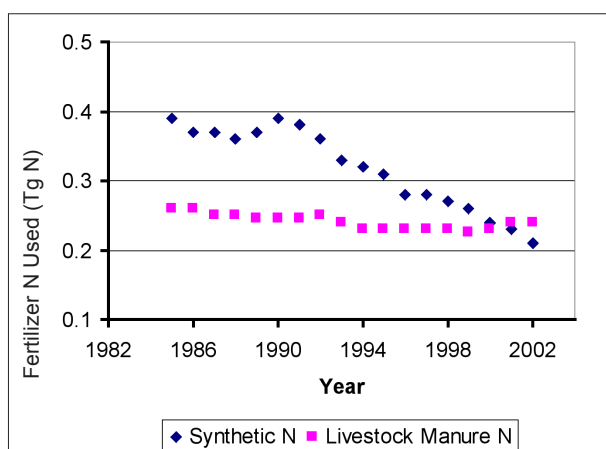


Figure 8: Synthetic fertilizer and livestock manure N used as fertilizer in Denmark 1985-2003

Source: IFA, 2004. Reprinted with permission from the International Fertilizer Industry Association.

response to the European Union Nitrate Directive. Synthetic fertilizer N use in Denmark decreased from approximately 0.4 Tg N in 1991 to 0.2 Tg N in 2002. Animal manure N application decreased from 0.25 Tg N to approximately 0.24 Tg N during this time period. Despite this large decrease in N fertilizer use, cereal crop yields remained relatively constant (data not shown). But while grain yields were maintained, there was a decrease in grain quality as determined by protein content (Figure 9). Grain protein content in wheat is critical for determining its quality for bread. In the U.S., a grain protein content of 12% is considered the threshold for good quality bread wheat, and N fertilizer application rate has a large influence on determining this trait (Cassman et al., 1992). As can be seen, grain protein content has declined from 12 to 10% in Denmark over the same period of lower fertilizer application rates.

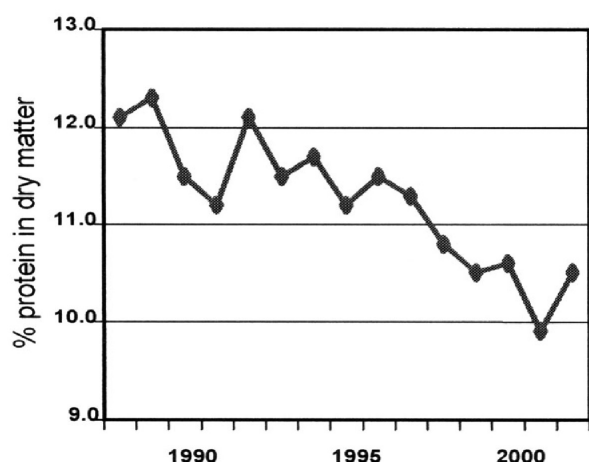


Figure 9: Protein content of cereal grain in Denmark

Source: IFA, 2004. Reprinted with permission from the International Fertilizer Industry Association.

Such trends raise several questions if declines continue or are found to be widespread. What is the cost to the farmer (considering human nutrition and end-use value costs)? Do these costs offset the environmental benefits created by decreasing N flows from crop production areas? What would be the regional and global impact if similar reductions in nitrogen fertilizer inputs to agriculture were put in place in developed countries that represent the largest source of grain exports to international markets?

As previously discussed, it is possible to increase crop yields without an increase in N fertilizer input if the methods used to apply N are modified to improve NFUE as illustrated by U.S. maize yield trends and fertilizer use (Figure 7). However, at some point continued yield increases will require additional N input because of the tight relationship between yield and crop N uptake requirements. The goal of N fertilizer management should therefore be to achieve highest possible NFUE while also sustaining crop yield increases adequate to meet food demand without need for a large expansion of crop production area. In fact, studies that evaluate NFUE in production agriculture find a wide range in NFUE among farmers. For farmers who achieve high NFUE, a reduction in N fertilizer rate would reduce yields. Hence, mandating an across-the-board reduction in N fertilizer use penalizes the good farmers who achieve high NFUE as their yields would likely fall while it would have little effect on yields of farmers who currently over fertilize (Cassman et al., 2003; Dobermann and Cassman, 2004).

Finally, if yields were significantly reduced as a result of lower N fertilization rates, more land may need to be brought into production. Because nearly all prime agricultural land is already used for crop production, expansion of crop area will most likely occur on more marginal land, such as the land currently in the Conservation Reserve Program (CRP). Such conversion would have two negative results. First, additional N losses from these acres would occur due to relatively low N fertilizer efficiency that typically occurs on marginal land that has multiple soil constraints to crop growth and yield. Second, conversion from CRP to crop production would result in loss of soil organic matter that accumulates under grassland, which in turn would lead to increased greenhouse gas emissions from agriculture.

Finding 2: N inputs to crop systems are critical to sustain crop productivity and soil quality. Moreover, given limited land and water resources, global population growth, and rapid economic development in the world's most populous countries, the challenge is to accelerate increases in crop yields on existing farm land while also achieving a substantial increase in N fertilizer uptake efficiency. This process is called "ecological intensification" because it recognizes the need to meet future food, feed, fiber, and energy demand of a growing human population while also protecting environmental

quality and ecosystem services for future generations (Cassman, 1999). More diverse cropping systems with decreased Nr fertilizer input may also provide an option to increase NFUE on a large scale if the decrease in Nr losses per unit of crop production in these diverse systems can be achieved without a decrease in total food production, which would trigger indirect land use change to replace the lost production and negate the benefits. However, crop cultivars and agronomic practices are changing rapidly, which changes N requirements, but current efforts in research, extension, and conservation programs on N management within these rapidly evolving systems are not adequate to meet the challenge of providing better information to increase NFUE.

Recommendation 2: *To obtain better information on Nr inputs and crop productivity, the Committee recommends that:*

Recommendation 2a: *Data on NFUE and N mass balance, based on direct measurements from production-scale fields, should be generated for the major crops to identify which cropping systems and regions are of greatest concern with regard to mitigation of Nr load and to better focus research investments, policy development, and prioritization of risk mitigation strategies.*

Recommendation 2b: *Efforts at USDA and universities should be promoted to: (i) investigate means to increase the rate of gain in crop yields on existing farm land while increasing N fertilizer uptake efficiency and (ii) explore the potential for more diverse cropping systems with lower N fertilizer input requirements so long as large-scale adoption of such systems would not cause indirect land use change.*

Recommendation 2c: *EPA should work closely with the USDA, Department of Energy, and the National Science Foundation, and universities to help identify research and education priorities to support more efficient use and better mitigation of Nr applied to agricultural systems.*

Biological fixation in cultivated croplands

Reactive nitrogen is also introduced to the landscape in significant quantities via BNF in cultivated crop lands. Management of biologically fixed N, insofar as it is possible, is proportionally as critical a task as the management of synthetic N because Nr from BNF is prone to the same loss pathways as Nr from commercial fertilizers. To quantify BNF due to human cultivation of crops, the Committee calculated the annual agricultural fixation for 2002 using crop areas and yields reported by the USDA Census of Agriculture (USDA, 2002). The Committee multiplied the area planted in leguminous crop species by the rate of N fixation specific to each crop type, assigning rates based on a literature review, as summarized in Table 4 below and shown relative to other inputs in Table 1. Annual nitrogen inputs to cropping system from BNF by legume crops was 7.7 Tg N/yr in 2002, accounting for ~15% of the overall Nr inputs to the terrestrial landscape from all sources and 20% of the agricultural sources (Table 1). Soybean and alfalfa contributions are the most important agricultural legumes in terms of nitrogen input and contribute 69% of total BNF inputs in U.S. agriculture.

Emissions factors and losses to the environment from fertilizers and organic nitrogen sources

Agriculture is a significant contributor of Nr inputs into the atmosphere. Nitrogen fertilizer losses vary greatly

Table 4: Estimates of nitrogen input from biological nitrogen fixation from major legume crops, hay, and pasture

Nr fixation in cultivated croplands				
	production area, Mha	rate, kg/ha/yr	Tg N/yr	% of total *
Soybeans	29.3	111	3.25	42
Alfalfa	9.16	224	2.05	27
Other leguminous hay	15.4	117	1.80	23
Western pasture	161	1	0.16	2
Eastern pasture	22.0	15	0.33	4
Dry beans, peas, lentils	0.88	90	0.08	1
Total			7.67	100

Source: Updated estimate for soybean based on a generalized relationship between soybean yield and the quantity of N fixation (Salvagiotti et al., 2008). Other values are from Boyer et al. (2002).

* Because of number rounding, the sum of individual percentages does not equal 100%.

due to differences in soil properties, climate, and the method, form, amount, timing and placement of applied N (Cassman et al., 2002). In addition, any factor that affects crop growth vigor and root system function also affects the ability of the plant to recover applied N efficiently. For example, denitrification can range from 0% to 70% of applied N (Aulakh et al., 1992). This process is mediated by heterotrophic, facultative anaerobic soil bacteria that are most active under warm, wet soil conditions; they have low activity in dry sandy soils.

Despite this variation, watershed, regional, and national assessments of carbon and N cycling often rely on average values for losses from each pathway. For example, the Intergovernmental Panel on Climate Change (IPCC) assumes that 1% of applied N fertilizer (uncertainty range of 0.3-3.0%) is lost from direct emissions of N₂O at the field level due to nitrification/denitrification. This assumption is based on analysis of all appropriate scientific publications that report these losses for specific crops and cropping systems (IPCC, 2007a). The same 1% default emission factor for field-level N₂O emission is applied to other N inputs from crop residues, organic amendments such as manure, and from mineralization of native soil organic matter. Data from scores of field studies were used to obtain this average value. A number of recent studies confirm that N₂O losses to the environment during the growing season at the field level can represent <1% of the applied nitrogen – even in intensive, high-yield cropping systems (Adviento-Borbe et al., 2006). Despite these average values, it is also clear that N₂O losses can vary widely even within the same field and from year to year due to normal variation in climate and crop management (Parkin and Kaspar, 2006; Snyder et al., 2007). Moreover, the loss of nitrogen from agricultural watersheds is strongly dependent on climate change (e.g., rainfall changes). Predicted increases and decreases in rainfall will likely have a dramatic impact on nitrogen export from agricultural fields. For example, precipitation is predicted to increase in the upper Mississippi watershed and, other factors being equal, N export should increase (e.g., Justic et al., 1995a,b).

Additional indirect N₂O emissions result from denitrification of volatilized NH₃ deposited elsewhere or from NO₃⁻ lost to leaching and runoff as the N_r cascades through other ecosystems after leaving the field to which it was applied. Here the IPCC assessment protocol assumes that volatilization losses represent 10% of applied N, and that N₂O emissions for these losses are 1% of this amount; leaching losses are assumed to be 30% of applied nitrogen and N₂O emissions are 0.75% of that amount (IPCC, 2007a). Therefore, the IPCC default value for total direct and indirect N₂O emissions represents about 1.4% of the applied N from fertilizer. By the same calculations, 1.4% of the N in applied organic matter, either as manure or compost or in

recycled crop residues, is also assumed to be emitted as N₂O. Recent work funded by EPA used the DAYCENT model to estimate N₂O emissions from cropping systems (Del Grosso et al., 2005). However, due to the cost of field validation of such models, there are relatively few validations across a representative range of cropping systems and environment. Therefore, it is not clear that use of such a complex model gives better estimates of N₂O emissions than the more straightforward IPCC assessment protocol.

Others have estimated higher average global N₂O losses of 3-5% of applied nitrogen fertilizer based on historical changes in atmospheric N₂O content and changes in N_r production during the past 50 years (Crutzen et al., 2008), as opposed to the field-based estimates that form the basis of IPCC estimates. Because N₂O is such a potent GHG, and given the more than two-fold difference in estimates of N₂O losses, there is a critical need to improve understanding and prediction of N₂O losses from agricultural systems. N₂O emissions in the U.S. are estimated to be 0.78 Tg N/yr (Table 5) (U.S. EPA, 2005b).

Table 5: N₂O emissions in the United States, 2002

	Tg N/yr	% *
Agricultural Soil Management	0.54	69
Manure Management	0.03	4
Mobile Combustion	0.09	12
Stationary Combustion	0.03	4
Nitric & Adipic Acid Production	0.05	6
Wastewater Treatment	0.02	2
Other	0.02	2
Total	0.78	100

* Because of number rounding, the sum of individual percentages does not equal 100%.

Biogenic NO_x emissions from croplands are on the order of 0.5% of fertilizer input – much more than this in sandy soils and less as clay content increases (Aneja et al. 1996; Sullivan et al. 1996; Veldkamp and Keller, 1997; Civerolo and Dickerson, 1998). However, NO_x emissions by agricultural burning are relatively unimportant. Ammonia volatilization of N from applied fertilizer can be the dominant pathway of N loss in rice soils and can account for 0% to more than 50% of the applied N depending on water management, soil properties, and method of application (citations within Peoples et al.,

1995, 2004). Ammonia volatilization can be of the same range in upland cropping systems, with largest losses occurring typically on alkaline soils (Peoples et al., 1995, 2004). The IPCC (2007a) uses a value of 10% of synthetic fertilizer N application and 20% of manure N as estimates of average NH_3 volatilization.

Taken together, N losses from all forms of direct gaseous emissions from crop production systems can represent a substantial portion of applied N fertilizer when soil conditions favor such emissions and there is a lack of synchrony between the amount of N applied and the immediate crop demand (Goulding, 2004). Therefore, achieving greater congruence between crop demand and the N supply from fertilizer is a key management tactic to reduce N losses from all sources. Success in reducing N losses and emissions from agriculture will depend on increased efforts in research and extension to close gaps in our understanding of N cycling and management in crop production, especially as systems further intensify to meet rapidly expanding demand for food, feed, fiber, and biofuel.

Finding 3: Nitrous oxide emissions from the Nr inputs to cropland from fertilizer, manure, and legume fixation represent a large proportion of agriculture's contribution to GHG emissions, and the importance of this source of anthropogenic GHG will likely increase unless NFUE is markedly improved in crop production systems. Despite its importance, there is considerable uncertainty in the estimates of nitrous oxide emissions from fertilizer and research should focus on reducing this uncertainty.

Recommendation 3: *The Committee recommends that EPA ensure that the uncertainty in estimates of nitrous oxide emissions from crop agriculture be greatly reduced through the conduct of EPA research and through coordination of research efforts more generally with other agencies such as USDA, DOE, NSF and with research conducted at universities.*

Impact of biofuel production capacity on Nr flux in agriculture

The enormous use of liquid fuels in the U.S., the rising demand for petroleum based liquid fuels from countries like China and India, and the decline in petroleum discovery all contributed to the recent record high petroleum prices. In addition, most of the world's petroleum reserves are located in politically unstable areas. This has provided strong motivation for policies promoting investment in biofuels made from corn, oil crops, and ultimately from cellulosic materials. In the U.S., ethanol production capacity from corn more than doubled from 2006 to 2009 (to a capacity of over 47 billion liters/year in January 2009). The renewable fuels standard in the 2007 Energy Independence and Security Act (EISA) will support another 9.5 billion liters/year of corn-based ethanol by 2015. An additional 79.5 billion liters is to come from cellulosic ethanol

by 2022. Production of biodiesel from vegetable oils also is encouraged in EISA, but expansion has been slowed by the high food value of such oils. Brazil is rapidly expanding its production of relatively low cost sugarcane ethanol and U.S. policies continue to be aimed at bringing about increased future biofuel production in the U.S.

In 2007 and 2008 petroleum prices pushed ethanol prices high enough to draw corn from food and feed uses into ethanol production and contribute to the increased price of corn. Because of the increase in petroleum/ethanol prices and the government subsidy for ethanol production, 30% of the corn crop was used to produce ethanol in 2008 (Abbott et. al., 2008). With the subsequent collapse in petroleum and ethanol prices, followed by corn prices, there has been unused capacity in the U.S. ethanol industry as the corn/ethanol price ratio made ethanol production uneconomic for some firms. However, EISA is likely to lead to the production of cellulosic materials and even some expanded corn production for biofuels once the U.S. gets beyond the current blending limit for ethanol (Doering and Tyner, 2009). The higher corn prices of 2007 and 2008 resulted in more land being planted to corn and higher N fertilizer requirements. Corn area went from 78.41 million acres (31.73 million hectares) in 2006-2007 to 93.6 million acres (37.88 million hectares) in 2007-2008. Reduction in soybean production accounted for 12.01 million acres (4.86 million hectares) of the corn expansion, and the remaining new acreage came primarily from reduced cotton acreage and from hayland and pasture. This strong response to high demand for biofuel feedstock has led to concern about increased pressure on the environment from biofuels. One important factor is the increased N necessary for growing corn and cellulosic materials (Robertson, et. al. 2008). Biofuels production and consumption will also result in NO_x emissions to the atmosphere. Expansion of corn or cellulosic materials production into marginal lands can be even more problematic with respect to nutrient leaching and soil erosion. Changes in N fertilizer prices add uncertainty to the additional amounts of N that may ultimately be used in biofuel feedstock production. Production of large amounts of distillers grains co-product is also changing the way that livestock feed rations are formulated, which in turn could have an influence on the cycling of N in cattle manure (Klopfenstein et al., 2008).

In February 2010, EPA released its final rule for the expanded Renewable Fuels Standard (RFS2) regarding greenhouse gas emissions from various types of biofuels based on life-cycle analysis as required by the 2007 EISA. Standards for corn ethanol and soybean biodiesel are based on studies that include data from large-scale production systems because these are the only biofuels currently produced on a large commercial

scale. In contrast, life cycle analysis (LCA) standards for cellulosic and other advanced biofuels were based on data from “pilot- and bench-scale” studies, or in many cases on hypotheses and rough estimates. While these initial estimates meet requirements imposed by 2007 EISA, the science underpinning life cycle assessments of biofuel systems, including direct and indirect land use change, are relatively undeveloped and evolving rapidly.

Finding 4: Rapid expansion of biofuel production has the potential to increase N fertilizer use through expanding corn production and its associated N fertilizer inputs and extending cultivation for cellulosic materials that will also need N. Distillers grains are changing animal diets and affecting N recycling in livestock. Both have important consequences for effective future management of Nr, but current models and understanding of how expanded biofuel production will affect Nr inputs and outputs from agriculture are inadequate to guide policy.

Recommendation 4: EPA should work with USDA and universities to improve understanding and prediction of how expansion of biofuel production, as mandated by the 2007 EISA, will affect Nr inputs and outputs from agriculture and livestock systems. Rapid expansion of biofuel production has the potential to increase N fertilizer use through expansion of corn production area and associated N fertilizer inputs, and from extending cultivation of cellulosic materials that will also need N inputs. Current models and understanding are not adequate to guide policy on how to minimize impact of biofuel expansion on environmental concerns related to Nr.

2.2.3. Nr Inputs and Losses from Animal Agriculture

In the U.S., domestic animals produce 6.0 Tg N/yr in manure and are the largest source of atmospheric $\text{NH}_3\text{-N}$ (1.6 Tg N/yr) (Table 1). Livestock also contribute to $\text{N}_2\text{O-N}$ emissions, though in much smaller proportions (~4% of total U.S. $\text{N}_2\text{O-N}$ emissions).

Trends in Animal Agriculture

While animal production has been increasing since World War II, this report emphasizes the period from 1970 to 2006. The production of chicken broilers increased more than four fold from 1970 to 2006 (Figure 10) and milk production increased by nearly 60% in this time period (Figure 11). Turkey production doubled and pork production increased about 25%, while meat from cattle (beef and dairy) remained constant (Figure 10).

Another trend in animal production has been for fewer animals to produce more animal products. For example, the 60% increase in the amount of milk produced in 2006 compared to 1970 required 25% fewer cows (Figures 11 and 12). Animal inventories declined by 10% for beef

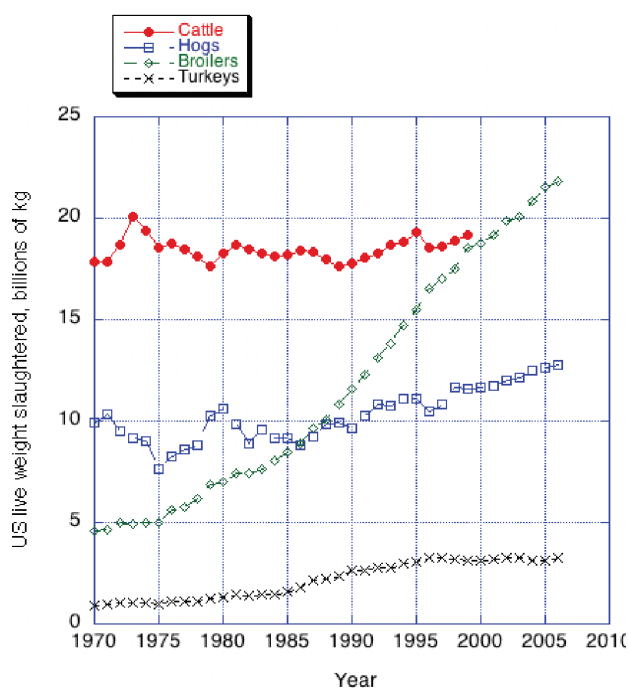


Figure 10: Meat production from 1970 to 2006

Source: USDA, NASS, 2007 – Census Reports. Data on cattle were not taken after 1999.

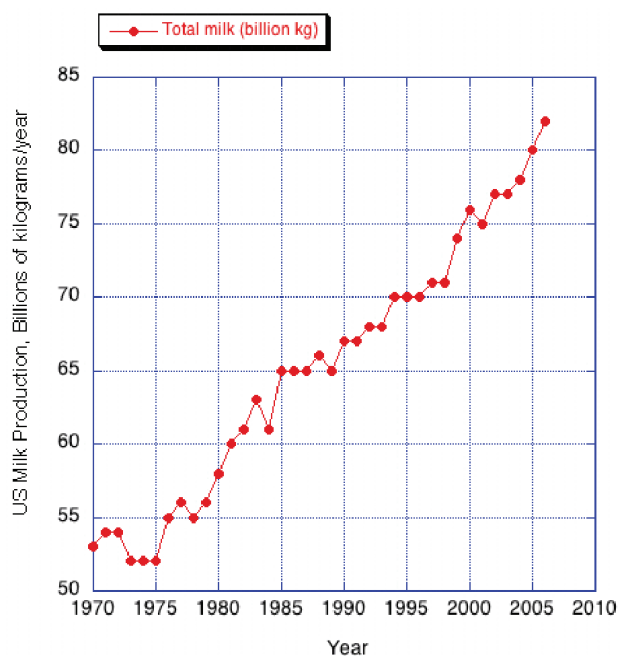


Figure 11: Milk production from 1970 to 2006

Source: USDA, NASS, 2007 – Census Reports

brood cows from 36 million head in 1970 to 33 million head in 2006, and the inventory of breeder pigs and market hogs declined 8% from 673 million head to 625 million head in the same period, even with similar or greater annual meat production. This trend resulted from

greater growth rates of animals producing more meat in a shorter amount of time. In 1970, broilers were slaughtered after 80 days on feed at 1.7 kg live weight, but by 2006 the average weight was 2.5 kg after only 44 days on feed (USDA, NASS, 2007).

Another trend in animal agriculture has been the increased size and smaller number of animal operations, which results from the mechanization of agricultural

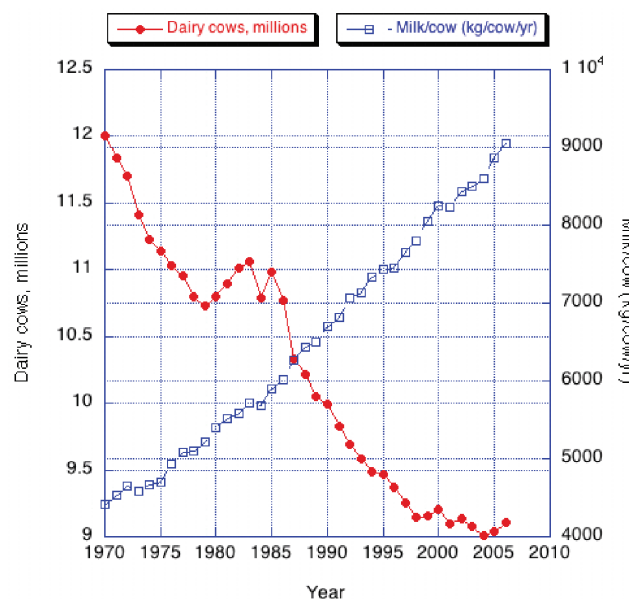


Figure 12: U.S. inventory of mature dairy cows and milk production per cow from 1970 to 2006

Source: USDA, NASS, 2007 – Census Reports

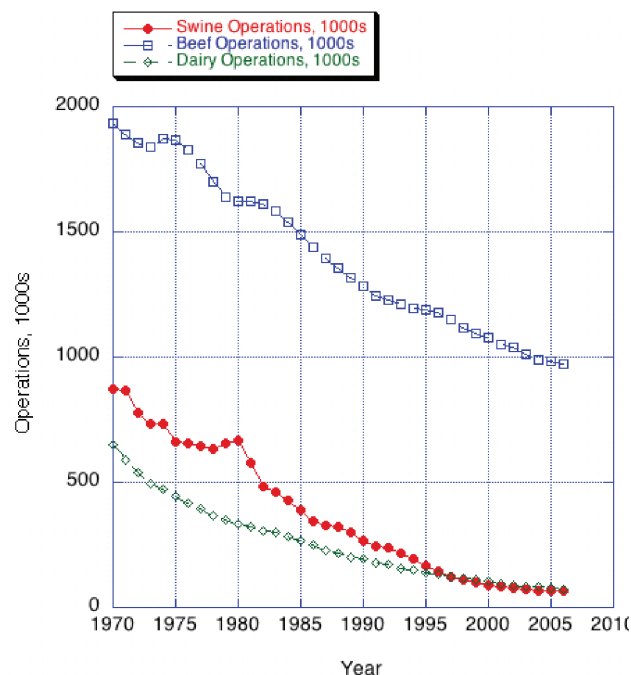


Figure 13: Number of animal operations in the United States from 1970 to 2006

Source: USDA, NASS, 2007 – Census Reports

practices and increased specialization. There were only 7% as many swine operations and 11% as many dairy operations in 2006 as there were in 1970 (Figure 13). There were half as many beef operations in 2006 as in 1970, but beef operations also expanded in size while smaller producers held jobs off the farm.

All of these trends show an increase in management and labor efficiency to produce a similar or greater amount of animal products. Also, because animal production is more concentrated on fewer farms with greater specialization, fewer crops are produced on those farms. As a result, it is increasingly common to have more manure nutrients produced on a livestock farm than can be used efficiently as fertilizer for crops on that farm. Therefore, unless the manure is applied over a larger crop area, the resulting over-application of manure on the livestock farm can reduce the subsequent efficiency of its utilization and result in greater nutrient losses.

Impact of livestock production trends on nitrogen use efficiency

The trends in livestock production have both positive and negative environmental impacts. One of the significant positive impacts is that with smaller animal inventories producing greater quantities of animal products, there is an improved efficiency of nitrogen utilization per product produced. This effect is partly the result of effectively reducing maintenance requirements during production. The requirements for feeding animals can be divided into two components: maintenance and production. The maintenance component is the feed that is used to keep the animal alive and healthy so that production is possible. The production component includes the feed that is converted to animal protein and waste due to the inefficiencies of these conversions. The maintenance component depends upon the number of animals, each animal's mass, and the time the animal is on feed. Thus, the maintenance requirement is diluted by faster growth rates and greater body weight at slaughter. The increases in production rates over time have led to greater efficiencies in N and P utilization for animal production and lower amounts of nutrients excreted per unit of animal protein produced.

Public concerns about the potential environmental and health effect of air emissions from CAFOs expand the impacts of food production beyond those associated with traditional agricultural practices (NRC, 2001; Aneja et al., 2009). Increased emissions of N compounds from animal agriculture into the atmosphere may lead to increased odor and interactions in atmospheric reactions (e.g., gas-to-particle conversion) (Baek et al., 2004a; Baek and Aneja, 2004b). These compounds are then transported by wind and returned to the surface by wet and dry deposition processes, which may have adverse effects on human health and the environment (McMurry et. al, 2004; Aneja et. al, 2006b, 2008a,b,c; Galloway et. al, 2008).

Adverse effects, further discussed in Chapter 3 of this report, include eutrophication, soil acidification, loss of biodiversity, and reactions that increase the mass concentration of atmospheric aerosols (PM_{2.5}). Aerosol formation occurs when HNO₃ reacts with basic compounds, and NH₃ reacts with compounds. Ecosystem acidification can occur when HNO₃ is deposited from the atmosphere. In addition, acidification can also occur when NH_x is deposited due to the production of HNO₃ from nitrification via soil microbes. Soil acidification occurs when HNO₃ or NH₄⁺ deposits on soils with low buffering capacity and this can cause growth limitations to sensitive plant species. Deposition of NO₃⁻ or NH₄⁺ also causes eutrophication (i.e., an over-abundance of nutrients), which can promote harmful algal growth leading to the decline of aquatic species. In fact, volatilized NO₃⁻ can travel hundreds of miles from its source, affecting local and regional biodiversity far from its origin (Aneja et al. 2008b; James, 2008).

The potential for reduced environmental impact from Nr in livestock systems depends on the proportion of the total intake attributable to maintenance costs. The commonly used tables for diet formulation published periodically by the NRC for various animal commodities can be used to track diet formulation practices and assumptions regarding maintenance and production requirements. About one third of the energy intake recommended for growing broilers was assumed to be needed for maintenance (NRC, 1994) but protein requirements were not divided between maintenance and production. For example, a dairy cow producing 40 kg milk per year would divert about 25% of its energy and 12% of its protein to maintenance (NRC, 2001).

In terms of nutritional efficiency of a herd or flock, maintenance of a productive phase (e.g., growth, lactation) also requires maintenance of a reproductive phase of the animal's life cycle. In other words, the actual nutritional maintenance cost of a herd or flock is greater than it is for productive individuals only. For example, milk production requires non-lactating cows and heifers in the herd which do not produce milk but which consume nutrients. These additional maintenance costs are lower for broiler flocks than for cattle.

Finding 5: There are no nationwide monitoring networks in the United States to quantify agricultural emissions of greenhouse gases, NO, N₂O, reduced sulfur compounds, volatile organic compounds (VOCs), and NH₃. Satellite observations of agricultural emissions hold promise for providing a high degree of spatial coverage and may complement surface observations in this network, although robust methods are yet to be developed. In contrast there is a large network in place to assess the changes in the chemical climate of the United States associated with fossil fuel energy production, i.e., the National Atmospheric Deposition Program/ National Trends Network (NADP/NTN), which has been

monitoring the wet deposition of sulfate (SO₄²⁻), NO₃⁻, and NH₄⁺ since 1978.

Recommendation 5: *The status and trends of gases and particulate matter precursors emitted from agricultural emissions, e.g., NO₃⁻ and NH₄⁺ should be monitored and assessed utilizing a nationwide network of monitoring stations. EPA should coordinate and inform its regulatory monitoring and management of reactive nitrogen with the multiple efforts of all agencies including those of the U.S. Department of Agriculture and NSF supported efforts such as the National Ecological Observatory Network (NEON) and the Long Term Ecological Research Network (LTER).*

Changes in feeding practices

From 1970 to 2006, several feeding practices were changed for diets fed to livestock (NRC, 1977, 1988, 1996, 2001). In 1989 and 1996, the NRC introduced the idea of dividing the form of protein fed to ruminants into that which is degraded by rumen microorganisms and that which passes through the rumen to be digested directly in the stomach and small intestine. Feeding ruminants with attention to rumen degraded and rumen undegraded protein decreases the amount of protein fed by 10 to 15% for a given protein requirement. For poultry and swine, manufactured amino acids were added to diets, decreasing the need for protein by 30%. Today, two amino acids (lysine and methionine), coated in a way to prevent degradation in the rumen, are sometimes added to dairy cattle diets thereby decreasing protein intake by another 15% (NRC, 2001). Phytase added to swine and poultry diets in the past decade has decreased phosphorus feeding by 20 to 50% with some of the decrease attributed simply to better understanding phosphorus requirements.

It is difficult to estimate the combined effects of these changes in feeding practices, but we have calculated changes in manure N by using USDA NASS data and assuming improvements in both production rates and ration formulation (Table 6). In the case of beef cattle diet formulation, the changes in feeding practices were determined by comparing the NRC 1976 recommendations with the NRC 1996 recommendations. Surprisingly, NRC 1996 recommended greater total crude protein compared to NRC 1976 despite formulating for rumen degraded and un-degraded protein and considering amino acid content. Therefore, improved diet formulation did not decrease N intake for beef in this time range but the effect of reduced maintenance did improve efficiency of N utilization.

Reduced nitrogen excretion from increased efficiency

Nitrogen excretion as fraction of animal production decreased from 1970 to 2006 (Table 6). However, in cases where the total amount of animal production in the U.S. increased substantially (e.g., broilers), total N excretion increased. The decrease in N excretion per unit of animal

Table 6: Livestock N excretion per kg production (g/kg) and per total United States (Tg/yr)

Commodity*	1970		2006	
	g/kg product	Total United States	g/kg product	Total United States
Milk	17	0.89	11	0.92
Pork, live weight	57	0.56	42	0.54
Broilers, live weight	56	0.26	46	1.00
Beef, live weight	123	1.2	110	1.3

*Does not include manure produced for reproduction of stock (e.g., growing dairy heifers, breeder pigs).

productivity was estimated by calculating the effects of changes in feeding practices and reduction of maintenance as described previously. The data in Table 6 indicate that there has been an increase in N utilization efficiency for livestock products.

For broilers, data are available to more accurately estimate the effect of changes in feeding and genetics on N excretion over time. However, these data do not represent the time period of interest in this report. Havenstein et al. (1994) compared a 1957 strain of broiler fed a 1957 diet to a 1991 strain fed a 1991 diet. Based on the reported N intake and production data, there was a 51% reduction in N excreted between these diets (Kohn, 2004).

Similarly, Kohn (2004) compared N excreted by U.S. dairy cows in 1944 and 2001. In 1944, the historically largest herd of dairy cattle in the U.S. (25 million cows) produced an average of 7 kg milk per cow per day (USDA NASS, 2007). In 2001, nine million cows produced an average of 27 kg milk per cow per day. Assuming the cows in 1944 and 2001 were fed according to popular feeding recommendations of the time, the N intakes were 360 and 490 g/day per cow, and N excretion rates (N intake minus N in milk) were 326 and 364 g/day per cow. Multiplying by the number of cows in the U.S., shows that total milk production increased 40% from 52 billion kg to 73 billion kg, while N excretion decreased 60% from 3.0 Tg N to 1.2 Tg N, respectively.

Table 7 provides information on manure production from animal husbandry in the U.S. For Table 7, manure N was calculated for all U.S. animal agriculture using data on animal production from the 2002 Census of Agriculture (USDA, 2002). For data on livestock production (cattle, calves, poultry, hogs, and pigs), manure was calculated by the methods of Moffit and Lander (1999), following the exact methods they had used to compute manure from the 1997 Census of Agriculture, but using the updated information from the 2002 Census of Agriculture. For data on production of manure from other animals (horses, goats, and sheep), the table uses

Table 7: Manure production from animal husbandry in the continental United States, Tg N per year 2002.

	Tg N/yr	%
Cattle & Calves	4.35	72
Poultry	0.94	16
Hogs & Pigs	0.53	9
Horses, Goats & Sheep	0.19	3
Continental United States*	6.02	100

* Because of number rounding, the sum of manure production by category does not equal the reported total for the continental U.S.

coefficients for manure excretion as a function of average animal weights and animal inventory, taken from Battye et al. (1994).

Volatilization of animal waste

Ammonia volatilization is highly variable and is influenced by the amount of total ammoniacal nitrogen (TAN), temperature, wind speed, pH, chemical and microbiological activities, diffusive and convective transport in the manure, and gas phase resistance in the boundary layer above the source (Arogo et al., 2006).

EPA estimates annual manure N excreted in livestock production in the U.S. for the *Inventory of U.S. Greenhouse Gas Emissions and Sinks* (U.S. EPA, 2007e). For the year 2002, these estimates indicate that a total of 6.8 Tg of N was excreted in livestock manure. Only a fraction of this N (~1.24 Tg) was recovered and applied directly as a nutrient source for crop production. Approximately 1.8 Tg N was transferred from the manure management systems, most likely by ammonia volatilization. Other loss vectors include leaching and runoff during treatment, and storage and transport before soil application. The remainder of the N was deposited in pastures and rangeland or in paddocks. This N is also

susceptible to movement into the atmosphere and aquatic systems or incorporation into soil organic matter. By a combination of BMPs and engineered solutions it may be possible to reduce the emissions and discharge of odors, pathogens, and nitrogen compounds from agricultural operations (Aneja et al., 2008b,d).

Total manure production reported in Table 7 in the contiguous U.S. was estimated using USDA's method and yields an estimate of 6.0 Tg N/yr; while EPA's GHG inventory method in Table 8 yields a total for the U.S. of 6.8 Tg N/yr in 2002. The "greenhouse gas" method suggests 13% higher manure N production. This difference highlights uncertainty in the calculations. Although the values in Table 8 include Alaska and Hawaii, the values in Table 7 do not; however, given the relatively small amount of livestock production in those states, that does not contribute substantially to the difference.

Finding 6: Farm-level improvements in manure management can substantially reduce Nr load and transfer. While the NPDES permitting process for CAFOs does include nutrient and manure management plans to limit the transport of land applied nutrients off farms, substantial off-farm transport of Nr still occurs, especially via atmospheric transport. There are currently very few incentives or regulations to decrease these transfers and loads despite the existence of management options to mitigate.

Recommendation 6: *A policy, regulatory, and incentive framework is needed and should be developed to improve manure management to reduce Nr load and ammonia transfer, taking into account phosphorus load issues.*

2.2.4. Nr Inputs to Residential and Recreational Turf Systems

Turf grasses cover 12.6-16.2 million ha across the continental U.S. (Milesi et al., 2005). The area under turf grass is roughly the size of the New England states

and occupies an area up to three times larger than that of *irrigated corn* (The Lawn Institute, 2007). The majority of this turf area (approximately 75%) is in residential lawns. About 80% of all U.S. households have private lawns (Templeton et al. 1998) that average 0.08 ha in size (Vinlove and Torla, 1995). Another approximately 15 % of total turf grass area is in low-maintenance parks and approximately 10% is in athletic fields and golf courses, which often receive higher levels of N application due to hard use conditions.

Supplemental N fertilization is often necessary to maintain healthy and aesthetically pleasing turf color, high shoot density, and the ability to resist and recover from stress and damage. Nitrogen also may be derived from atmospheric deposition or recycled decomposition of soil and grass clipping organic matter. Whether these inputs are sufficient to maintain lawns of adequate quality depends on many factors including age of the turf, uses, and expectations or goals of the homeowner or field manager. Also, turf grasses are used to stabilize soil, often with an erosion-prevention matrix such as organic mats or with hydroseeding. Depending on circumstances, these turf uses may be temporary until natural vegetation succeeds the turf, or may be low-maintenance turfs that are seldom fertilized, such as highway medians and shoulders, grassy swales and buffers.

Turf grass is maintained under a variety of conditions. Approximately 50% of all turf grass is not fertilized; the remainder is fertilized at varied intensities (A.M. Petrovic, personal communication, June 5, 2007). The Committee has arrayed the different turf managements into the following three groups according to the estimated amount of N-fertilizer applied annually (Table 9): residential lawns maintained by homeowners (0.73 kg/100 m²), residential lawns cared for by professional lawn care companies (2.92 [range, 1.95-7.3] kg/100 m²), and athletic fields and golf courses (3.89 [range, 2.64-6.64] kg/100 m²). The estimate of total N-fertilizer used

Table 8: Fate of livestock manure nitrogen (Tg N)

Activity	1990	1992	1994	1996	1998	2000	2002	2004
Managed manure N applied to major crops	1.1	1.2	1.2	1.2	1.2	1.3	1.2	1.3
Manure N transferred from management systems	1.5	1.6	1.6	1.6	1.7	1.7	1.8	1.7
Pasture, range, & paddock manure N	3.9	4.0	4.1	4.2	3.9	3.8	3.8	3.7
Total *	6.6	6.7	6.9	7.0	6.9	6.8	6.8	6.7

Source: U.S. EPA, 2007e

* Because of number rounding, the sums of manure production do not always equal the reported totals.

Table 9: Estimate of fertilizer N used on turf grass in the United States in the year 2000, based on a total area of 12.6 million ha

Type of Turf Fertilized	Area (Million ha)	N rate (kg/ha/yr)*	Total N Used (Tg/yr)
Nominal Fertilization	4.7	73	0.35
Professional Lawn Care	0.93	296 (195-488)	0.27
High Maintenance Areas (golf/sports)	1.26	390	0.49
Total	6.89	–	1.11

Source: Data derived from Milesi; et al., 2005 and A.M. Petrovic, Personal Communication 2007

*A conversion factor of 1000 m²/ha was used with application rates (kg N/100 m²) of: 0.73 for nominal fertilization, 2.92 for professional lawn care, and 3.89 for high maintenance areas kg N/100 m²

on turf grass in the U.S. is 1.1 Tg N/year, or 9% of the total average annual N-fertilizer used between 1999 and 2005. Depending on land use patterns, certain areas of the country, particularly coastal areas where residential and urban properties prevail, turf fertilizer can be an important or even dominant source of nitrogen to surface waters.

Turf fertilizer N is susceptible to losses to the atmosphere and surface and ground water when it is not properly managed. Research on lawns has shown that leaching of NO₃⁻ can range between 0 and 50% of N applied (Petrovic, 1990). Nitrogen leaching losses can be greatly decreased by irrigating lightly and frequently, using multiple and light applications of fertilizers, fertilizing at the appropriate times (especially not too late in the growing season), and using soil tests to ensure proper balance of non-N soil condition and pH. In a soil column experiment with turf coverage, the percentage of N leached (as percentage of N applied) varied from 8% to 14% using light irrigation and from 2% to 37% with heavy irrigation. Applying fertilizer in appropriate amounts, avoiding periods when grass is dormant, and not fertilizing too soon before irrigation or large rainfall events can all help ensure leaching and runoff will be minimal without affecting turfgrass color and growth (Mangiafico and Guillard, 2006).

Nitrogen runoff losses are poorly quantified but a range similar to leaching is probable (A.M. Petrovic, personal communication). The chemical form of fertilizer N does not impact leaching/runoff unless the fertilizer is applied in late autumn (Petrovic, 2004a), although use of slow release or organic fertilizers can help reduce runoff and leaching. Shuman (2002) notes that runoff can be limited by applying minimum amounts of irrigation following fertilizer application and avoiding application before intense rain or when soil is wet. Losses of Nr to the atmosphere can be significant when urea is applied. Measured denitrification losses are usually small, but depend upon timing of N application relative to soil water status, irrigation, and temperature. Typically 25% of N applied is not accounted for in runoff, leaching, and uptake/removal, or soil

sequestration (A.M. Petrovic, personal communication). This suggests that volatilization and denitrification are important loss vectors. Nitrogen volatilization (Beard and Kenna, 2008) rates range from 0.9% under light irrigation to 2.3% under heavy irrigation.

While under-fertilization can lead to reduced grass stand and weed encroachment which results in more leaching and runoff N losses than from well managed lawns (Petrovic and Larsson-Kovach, 1996; Petrovic, 2004b), Guillard (2008) recommends not fertilizing lawns of acceptable appearance. Further, prudent fertilization practices may include using one-third to one-half (or less) of the recommended application rate (i.e., application rates below 0.5 kg/100m²) and monitoring response (Guillard, 2008). Less or no fertilizer may produce acceptable lawns, especially once the lawn has matured, provided clippings are returned and mowing length is left high.

As noted above, according to Petrovic (personal communication) half the lawns in the U.S. may not receive any fertilizer. Those lawns are presumably satisfactory to their owners. Further N reductions can be made if white clover is incorporated into turf, and grasses such as fescues (which require little or no N supplements once mature) are selected for amenable parts of the country. These practices can potentially reduce N fertilization (and subsequent leaching risk) on turf by one third or more, saving 0.4 or more Tg N/year. When properly managed, turf grass provides a variety of services that include decreasing runoff, sequestering carbon dioxide, and providing a comfortable environment in which to live (Beard and Green, 1994).

Finding 7: Synthetic N fertilizer application to urban gardens and lawns amounts to approximately 9% of the total annual synthetic N fertilizer used in the United States. Even though this N represents a substantial portion of total N fertilizer use, the efficiency with which it is used receives relatively little attention.

Recommendation 7a: *To ensure that urban fertilizer is used as efficiently as possible, the Committee*

recommends that EPA work with other agencies such as USDA as well as state and local extension organizations to coordinate research to ensure that fertilization recommendations are accurate and promote awareness of the issue.

Recommendation 7b: *Through outreach and education, supported by research, improved turf management practices should be promoted, including improved fertilizer application and formulation technologies and maintenance techniques that minimize supplemental Nr needs and losses, use of alternative turf varieties that require less fertilization, alternative ground covers in place of turf, and use of naturalistic landscaping that focuses on native species.*

2.3. Nr Transfer and Transformations in and Between Environmental Systems

This section discusses the transfers and flows of Nr within and between environmental systems (ES), which include atmospheric, terrestrial, and aquatic environments. The first Section (2.3.1) contains information on Nr deposition from the atmosphere to terrestrial and aquatic systems, presents estimates of input and recycling of Nr within terrestrial systems, and discusses movement of Nr from the terrestrial to the aquatic system. The second Section (2.3.2) presents an estimate of storage of Nr within the terrestrial system. Areas of uncertainty in Nr transfer and transformation are discussed in Section 2.3.3. Section 2.3.3 also contains an example analysis of Nr input and fate in 16 watersheds in the northeast U.S. No comprehensive national data are available to assess the transfer and transformations of Nr in and between the atmosphere, terrestrial systems and aquatic systems. The example analysis in Section 2.3.3 shows how an evaluation of inputs and fate of Nr could be conducted for a large watershed.

2.3.1. Input and Transfers of Nr in the United States

Nitrogen deposition from the atmosphere to the earth's surface

Atmospheric input contributes substantially to the Nr content of terrestrial and aquatic ecosystems for the U.S., but the magnitude and mechanisms of Nr deposition to the earth's surface remain major unanswered environmental questions. Along the eastern U.S. coast and eastern Gulf of Mexico, atmospheric deposition of N currently accounts for 10% to over 40% of new N loading to estuaries (Paerl et al., 2002). Other watershed contribution estimates range widely throughout the U.S., depending on size of the watershed, the size of the estuary, and the magnitude of contributing sources of atmospheric N enrichment. Valigura et al. (2001) identified a median atmospheric N contribution of about 15% for 42 watersheds located throughout the U.S., although the maximum estimate was 60%.

NO_x , NH_3 and their reaction products not deposited onto continents are generally lofted into the free troposphere where they can have a wide range of influence and, in the case of NO_x , because of nonlinearities in the photochemistry, generate substantial amounts of tropospheric ozone (U.S. EPA, 2006a). Total N deposition involves both gases and particles, and both dry and wet (in precipitation) processes. Rates of deposition for a given species (in units of mass of N per unit area per unit time) can be measured directly, inferred from mass balance of the atmospheric budget, or modeled numerically, but substantial uncertainties remain with each of these techniques when applied to deposition of any Nr species. A portion of the Nr deposited to the earth's surface is re-emitted as NH_3 , NO, or N_2O (Galbally and Roy, 1978; Kim et al., 1994; Civerolo and Dickerson, 1998; IPCC, 2007a; Crutzen et al., 2008). Although naturally-produced Nr is involved, anthropogenic Nr dominates over most of the U.S. In Appendix A we provide a review of the state of the science concerning the total annual Nr deposition and trends in that deposition to the contiguous 48 states.

Deposition involves both oxidized and reduced N species. Of the oxidized forms of atmospheric N, all the members of the NO_y family (NO, NO_2 , NO_3 , N_2O_5 , HONO, HNO_3 , NO_3^- , PAN and other organo-nitrates, RONO_2) can be transferred from the troposphere to the surface, and some (e.g., NO) undergo bidirectional flux. Volatile amines are also detected as NO_y compounds (Kashihira et al., 1982; Wyers et al., 1993). Although a potent GHG, N_2O is only emitted, not deposited and therefore has not been considered in the material presented here and in Appendix A. Of the reduced forms of atmospheric nitrogen, NH_3 and NH_4^+ play a major role. There is also evidence of deposition of organic N such as amino acids and isoprene nitrates, and recent observations suggest that these can account for ~10% (possibly as much as 30%) of the U.S. NO_x budget, especially in summer (Keene et al., 2002; Horowitz et al., 2007; Duce et al., 2008; Sommariva et al., 2008). While this is a worthy research topic, measurements are still limited and deposition of organic N compounds has not been reviewed in this report. The wide array of relevant atmospheric compounds makes direct measurement and accurate load quantification challenging. Appendix A provides a discussion of wet and dry deposition of Nr in the U.S. and the relationship between emissions of Nr and observed deposition.

Consideration of NO_y as a supplement for the current NO_2 National Ambient Air Quality Standard

The six principal (or criteria) pollutants for which EPA has established National Ambient Air Quality Standards (NAAQS) include “oxides of nitrogen” (the sum of NO and NO_2) or NO_x . The specific chemical compound nitrogen dioxide (NO_2) has been selected as

the indicator for compliance with the NAAQS for NO_x. The levels of primary and secondary standards for NO₂ are identical at 0.053 ppm (approximately 100 µg/m³) in annual arithmetic average, calculated from the one-hour NO₂ concentrations. In forming an integrated policy for protecting the environment from adverse effects of reactive nitrogen, it is appropriate to consider whether the existing criteria pollutants are sufficiently inclusive of Nr species. EPA's recent Integrated Science Assessment for Oxides of Nitrogen and Sulfur – Ecological Criteria (U.S. EPA, 2008d) evaluated the scientific foundation for the review of the secondary (welfare-based) NAAQS for oxides of nitrogen and concluded that:

The instrumentation deployed at present in the routine monitoring networks for determination of gas-phase NO₂ and SO₂ concentrations is likely adequate for determining compliance with the current NAAQS. But in application for determining environmental effects, all these methods have important limitations, which make them inadequate for fully characterizing the state of the atmosphere at present, correctly representing the complex heterogeneity of N and S deposition across the landscape, and for realistically apportioning the contributions of reduced and oxidized forms of atmospheric N and S in driving observed biological effects at a national scale.

Although the current standard for NO₂ is inadequate to protect welfare, there is a straightforward technical fix to this problem. NO_x (NO + NO₂) is a variable, often small component of reactive oxidized nitrogen (NO_y) as has been noted in EPA's Integrated Science Assessment for Oxides of Nitrogen and Sulfur – Ecological Criteria and extensively documented in the reviewed literature (Fahey et al., 1986; Fehsenfeld et al., 1987; Doddridge et al., 1991; Parrish et al., 1993; Poulida et al., 1994; Ridley et al., 1994; Emmons et al., 1997; Liang et al., 1998; Munger et al., 1998; Zhou et al., 2002; Takegawa et al., 2003; Parrish et al., 2004a,b; Horii et al., 2004, 2006; Dunlea et al., 2007; Kleinman et al., 2007; Hargrove and Zhang, 2008; Luria et al., 2008; Zhang et al., 2008; Schwab et al., 2009; Luke et al., 2010). The standard chemiluminescence NO/NO_x technique measures NO directly via reaction with O₃, and NO_x (NO + NO₂) by conversion of NO₂ to NO on hot molybdenum. It had been thought that NO_x would make up nearly all of NO_y in heavily polluted urban areas, but results including some from Mexico City (Dunlea et al., 2007) indicate that this is not the case. Interferences were up to 50% of the ambient NO₂ concentration even in a heavily urban area. Recent results from Houston, TX (Luke et al., 2010) showed a median NO_x/NO_y ratio of ~0.63 between 1300 and 1500 local time, a time of rapid ozone production. NO₂ has been used historically as the indicator for many health studies, and it is regulated in part because of studies that directly link it to respiratory effects. However, given the inadequacy of NO₂ as a standard to protect welfare, the Committee recommends that EPA

reexamine the criteria pollutant oxides of nitrogen and the indicator species NO₂ and consider adding chemically reactive nitrogen as a criteria pollutant, and NH_x and NO_y as indicators to supplement the NO₂ National Ambient Air Quality Standard.

The references listed above also document that current suite of NO_x monitors are useful for proving compliance with the current NO₂ standard, but suffer substantial interferences and the extent of these interferences varies with time and location. Reactive N compounds, especially HNO₃, can be lost on the inlet components. Because the inlet does not transmit all reactive nitrogen compounds with high efficiency, commercial NO_x monitors may provide an upper limit for NO_x and a lower limit for NO_y, but they measure neither NO_x nor NO_y precisely. Numerical simulations do not currently produce consistent results for the partitioning of NO_y species, for example see Archibald et al., (2010). The data from the current monitoring network are of limited value for determining exposure to NO₂, for evaluating chemical transport models (CTMs), or for assessing efficacy of emissions control strategies (McClenny et al., 2002).

NO_y monitors differ from the NO_x monitors currently in use in the position of the hot molybdenum NO₂ to NO converter and in calibration and operation (e.g., Thermo Scientific Model 42i-Y). In an NO_x monitor the converter is well downstream of the inlet, behind a filter that removes particulate matter, while in an NO_y monitor the converter is at the inlet (Fehsenfeld et al., 1987; Doddridge et al., 1991; Doddridge et al., 1992; Parrish et al., 1993; Poulida et al., 1994; Ridley et al., 1994; Emmons et al., 1997; Liang et al., 1998; Munger et al., 1998; Parrish et al., 2004b; Dunlea et al., 2007; Hargrove and Zhang, 2008; Luria et al., 2008; Schwab et al., 2009). Thus the current monitors could be replaced or retrofitted to measure NO_y with a detection limit of 0.1 ppb at reasonable expense for equipment and for training operators. NO_y monitors, like NO_x monitors, provide proof of compliance with NO₂ standards. For specific measurement of NO₂, commercial instruments for selective NO₂ reduction (e.g., Air Quality Design, Inc., Wheat Ridge, CO) or direct NO₂ measurement are beginning to be available (Parrish and Fehsenfeld, 2000; Castellanos et al., 2009) and these may offer an alternative approach. Direct monitoring of NO₂ and other Nr species is a long-term goal, but new techniques will require thorough testing in polluted environments. NO_y monitors using hot molybdenum compare favorably with other NO_y techniques such as gold-catalyzed CO reduction and several studies have concluded that NO_y can be reliably measured in suburban and urban environments (Fehsenfeld et al., 1987; Crosley, 1996; Williams et al., 1998).

Conclusions on atmospheric deposition of Nr

As discussed here and in Appendix A, downward transport from the atmosphere is a major source of Nr to the earth's surface, but there are uncertainties in

the characteristics and absolute magnitude of the flux. Pollutants not deposited are exported from the continent and alter the composition and radiative balance of the atmosphere on a large scale. A review of the literature revealed the following major points concerning the present state of the science:

1. Measurements from the National Atmospheric Deposition Program (NADP) indicate that wet deposition of ammonium plus nitrate for the period 2000-2006 averaged 3.1 kg N/ha/yr over the 48 contiguous states.
2. The reduced (NH_4^+) and oxidized (NO_3^-) forms of reactive N contributed about equally to the flux, but input to the eastern United States was greater (and less uncertain) than to the western U.S.
3. For the U.S. east of the Mississippi River, dry deposition data have also been analyzed – the Clean Air Standards and Trends Network (CASTNET) monitors vapor phase HNO_3 , as well as particulate NO_3^- and NH_4^+ . These measurements indicate 7.75 kg N/ha/yr total deposition (5.46 wet 2.29 dry) over the East. Conspicuous by its absence from this number is dry deposition of ammonia.
4. Decreases in NO_x emissions appear to have led to decreases in NO_3^- deposition. NADP data show a national decreasing trend in the wet nitrate deposition and some individual sites show statistically significant decreases in deposition and correlations with emissions.
5. A thorough review of all published studies of the U.S. NO_y budget indicates that about 70 % of the NO_x emitted by the U.S. is deposited onto the continent with the remainder exported, although substantial uncertainty remains. Major sources of uncertainty concerning the deposition of NO_y include dry deposition of unmonitored members of the NO_y family, uncertainties in the chemistry of organic N, and poorly constrained estimates of convective venting of the planetary boundary layer of the atmosphere (i.e., uncertainties concerning the venting of NO_y from the lowest layer of the atmosphere).
6. Based on observations and model estimates of the relative deposition of unmeasured quantities, total estimated deposition of all forms of Nr for the period 2000-2004 is ~11 kg N /ha /yr for the eastern U.S., and for the 48 states ~7.5 kg N /ha /yr with a range of 5.5 to 9.5 kg N /ha /yr.

Finding 8: Scientific uncertainty about the origins, transport, chemistry, sinks, and export of Nr remains high, but evidence is strong that atmospheric deposition of Nr to the earth's surface as well as emissions from the surface to the atmosphere contribute substantially to environmental and health problems. Nitrogen dioxide, NO_2 , is often a small component of NO_y , the total of oxidized nitrogen in the atmosphere. The current NAAQS for NO_2 , as an

indicator of the criteria pollutant “oxides of nitrogen,” is inadequate to protect health and welfare. NO_y should be considered seriously as a supplement or replacement for the NO_2 standard and in monitoring. Atmospheric emissions and concentrations of Nr from agricultural practices (primarily in the form of NH_3) have not been well monitored, but NH_4^+ ion concentration and wet deposition (as determined by NADP and NTN) appear to be increasing, suggesting that NH_3 emissions are increasing. Both wet and dry deposition contribute substantially to NH_x removal from the atmosphere, but only wet deposition is known with much scientific certainty. Thus consideration should be given to adding these chemically reduced and organic forms of Nr to the list of Criteria Pollutants.

Recommendation 8a: EPA should reexamine the criteria pollutant “oxides of nitrogen” and the indicator species NO_2 and consider adding chemically reactive nitrogen as a criteria pollutant, and NH_x and NO_y as indicators to supplement the NO_2 National Ambient Air Quality Standard.

Recommendation 8b: Monitoring of NH_x and NO_y should begin as soon as possible to supplement the existing network of NO_2 compliance monitors.

Recommendation 8c: EPA should pursue the longer-term goal of monitoring individual components of Nr, such as NO_2 (with specificity), NO and PAN, and HNO_3 , and other inorganic and reduced forms, as well as support the development of new measurement and monitoring methods.

Recommendation 8d: The scope and spatial coverage of the Nr concentration and flux monitoring networks (such as the National Atmospheric Deposition Program and the Clean Air Status and Trends Network) should be increased and an oversight review panel for these two networks should be appointed.

Recommendation 8e: EPA in coordination with other federal agencies should pursue research goals including:

- Measurements of deposition directly both at the CASTNET sites and in nearby locations with non-uniform surfaces such as forest edges.
- Improved measurements and models of convective venting of the planetary boundary layer and of long range transport.
- Improved analytical techniques and observations of atmospheric organic N compounds in vapor, particulate, and aqueous phases.
- Increased quality and spatial coverage of measurements of the NH_3 flux to the atmosphere from major sources especially agricultural practices.
- Improved measurement techniques for, and numerical models of NO_y and NH_x species (especially with regard to chemical transformations, surface deposition and offshore export, and linked ocean-land-atmosphere models of Nr).

Input and recycling of Nr within terrestrial systems in the United States

Annual input of newly created Nr onto terrestrial ecosystems comes primarily from atmospheric deposition, synthetic fertilizer and BNF in managed and unmanaged ecosystems (Table 1). Although Nr deposited from the atmosphere to terrestrial systems is formed inadvertently during fossil fuel combustion and from volatilization of NH_3 from agricultural activities, it serves to provide nutrients, along with biological N fixation and synthetic fertilizer, for food, feed, and fiber production in the agricultural sector. Forests and grasslands use Nr for growth. Home gardens, parks and recreational areas utilize Nr within the urban landscape. Approximately 32 Tg of new Nr reached the land of the 48 contiguous states in 2002 (Table 1). An additional ~0.2 Tg of N was imported mainly as food and drink products (FAO, 2007). An additional ~12 Tg of Nr was recycled back to terrestrial and aquatic systems in livestock excreta (~6 Tg N), human excreta (~2 Tg N), and crop residue from the previous year's production (~4 Tg N; U.S. EPA, 2007e). Of this N, ~1.3 Tg (~1.2 from livestock manure and <0.1 from sewage sludge) was used as fertilizer for crop production (U.S. EPA, 2007e). More detailed information describing sources and cycling of reactive nitrogen input in terrestrial systems is provided in Appendix B.

Finding 9: Total N budgets within all terrestrial systems are highly uncertain. The relative amount of N losses from terrestrial systems ascribed to leaching, runoff, and denitrification are as uncertain as the N budgets themselves.

Recommendation 9: EPA should join with USDA, DOE, and universities to work together in efforts to ensure that the N budgets of terrestrial systems are properly quantified and that the magnitudes of at least the major loss vectors are known.

Transfer of Nr to aquatic systems

Within the nitrogen cascade, Nr flows from the atmosphere and terrestrial systems into aquatic systems. Aquatic systems include groundwater, wetlands, streams and rivers, lakes and the coastal marine environment. Nr is deposited directly into surface aquatic systems from the atmosphere (direct deposition) and Nr that is not either stored or removed as products on terrestrial systems eventually moves into aquatic systems (indirect deposition).

The area of an airshed generally greatly exceeds that of a watershed for a specific estuary or coastal region. For example, the airshed of the Baltic Sea includes much of western and central Europe (Asman, 1994; Hov et al., 1994), while the airsheds of the two largest U.S. estuarine ecosystems, the Chesapeake Bay and Albemarle-Pamlico Sound, are 15 to over 30 times the size of their watersheds (Dennis, 1997). Thus, the airshed

of one region may impact the watershed and receiving waters of another, making eutrophication a regional-scale management issue (Galloway and Cowling, 2002; Paerl et al., 2002). Furthermore, atmospheric N inputs do not stop at coastal margins. Along the North American Atlantic continental shelf, atmospheric N inputs more than match riverine inputs (Jaworski et al., 1997; Paerl et al., 2002), underscoring the fact that N-driven marine eutrophication may require regional or even global solutions. Even in truly oceanic locations (e.g., Bermuda), North American continental atmospheric N emissions (reduced and oxidized N) are commonly detected and significant (Luke and Dickerson, 1987; Prospero et al., 1996). Likewise, islands in the North Pacific receive N deposition originating in Asia (Prospero et al., 1989).

Riverine and atmospheric “new” Nr inputs in the North Atlantic Ocean basin are at least equal and may exceed “new” Nr inputs from biological N_2 fixation (Howarth et al. 1996; Paerl and Whitall, 1999; Paerl et al. 2002). Duce et al. (2008) estimate that up to one-third of the oceans’ external Nr supply enters through atmospheric deposition. This deposition leads to an estimated ~3% of new marine biological production and increased oceanic N_2O production. Schlesinger (2009) estimated that global atmospheric transport of Nr from land to sea accounts for the movement of almost one third of the annual terrestrial Nr formation. Therefore, our understanding of marine eutrophication dynamics, and their management, needs to consider a range of scales reflecting these inputs, including ecosystem, watershed, regional and global levels.

One example of shifting N inputs is the proliferation of intensive livestock operations in coastal watersheds, which has led to large increases and changes in chemical composition of nitrogenous compounds discharged to estuarine and coastal waters via runoff, groundwater, and atmospheric deposition (Paerl, 1997; Howarth, 1998; Galloway and Cowling, 2002). In general, coastal waters under the influence of these operations are experiencing increases in total N loading as well as a shift toward more reduced N (NH_4^+ , organic N) relative to oxidized N (NO_3^-) (Galloway and Cowling, 2002; Howarth et al., 2002). These increases, combined with increases in hypoxia and anoxia in receiving waters, are leading to more NH_4^+ -rich conditions, which will favor those algal groups best able to exploit this N form, including some harmful algal bloom (HAB) taxa (Paerl and Whitall, 1999; Paerl et al., 2007). Similarly, conversion of forest and agricultural lands to urban lands can alter landscapes and promote N loading to estuaries by increasing impervious pathways and removing natural landscape filters for Nr. Development also destroys wetlands, leading to more NO_3^- -enriched conditions, potentially favoring plant taxa best able to exploit this N form.

A recent evaluation of decadal-scale changes of NO_3^- concentrations in ground water supplies indicates

Box 1: Hypoxia in the Gulf of Mexico

An example of a problem of excess Nr that moves from one part of the U.S. to another is the movement of Nr from the states that make up the Mississippi River drainage to the Gulf of Mexico (see discussion in Section 5.3.4). A hypoxic zone covers a significant area of the receiving bottom waters of the continental shelf of the northern Gulf of Mexico (details may be gleaned from U.S. EPA SAB, 2007). This is a seasonally severe problem that has persisted there for at least the past 20 years. Between 1993 and 1999 the hypoxia zone ranged in extent from 13,000 to 20,000 km² (Rabalais et al. 1996, 1999; Rabalais and Turner, 2001). The hypoxia is most widespread, persistent, and severe in June, July, and August, although its extent and timing can vary, in part because of the amplitude and timing of flow and subsequent nutrient loading from the Mississippi River Basin. The waters that discharge to the Gulf of Mexico originate in the watersheds of the Mississippi, Ohio, and Missouri Rivers (collectively described here as the Mississippi River Basin). With a total watershed of 3 million km², this basin encompasses about 40% of the territory of the lower 48 states and accounts for 90% of the freshwater inflow to the Gulf of Mexico (Rabalais et al. 1996; Mitsch et al., 2001; U.S. EPA SAB, 2007).

The December, 2007 report, *Hypoxia in the Northern Gulf of Mexico: An update by the EPA Science Advisory Board* (U.S. EPA SAB, 2007) determined that, “To reduce the size of the hypoxic zone and improve water quality in the Basin, the SAB Panel recommends a dual nutrient strategy targeting at least a 45% reduction in riverine total nitrogen flux (to approximately 870,000 metric tons/yr) and at least a 45% reduction in riverine total phosphorus flux (to approximately 75,000 metric tons/yr). Both of these reductions refer to changes measured against average flux over the 1980-1996 time period. For both nutrients, incremental annual reductions will be needed to achieve the 45% reduction goals over the long run. For nitrogen, the greatest emphasis should be placed on reducing spring flux, the time period most correlated with the size of the hypoxic zone.”

that there has been a significant increase in nitrate concentrations in well water across the U.S. (Rupert, 2008). This study compared the nitrate content of 495 wells during 1988-1995 with nitrate content found during 2000-2004 as a part of the U.S. Geological Survey, National Water Quality Assessment Program. In a subset of wells, ground water recharge was correlated with historic fertilizer use and it was concluded that nitrate concentrations in ground water increased in response to the increase of N fertilizer use.

2.3.2. Storage of Nr within Terrestrial Environmental Systems

According to the nitrogen cascade conceptualization, terrestrial environmental systems are compartmentalized into agriculture, populated, and vegetated systems. Annual input of Nr is greatest in agricultural ecosystems (farmland, cropland, and grazed pastureland). Annual Nr inputs to agricultural ecosystems, using 2002 as the base year, include 9.8 Tg from synthetic fertilizer, 7.7 Tg from biological N fixation in crops (mainly soybeans), and 1.3 Tg from atmospheric deposition. Nr input into vegetated systems (mostly forested, but including non-cropland grasslands and other natural vegetation types as well) comes mostly from atmospheric deposition (3.2 Tg). Annual input of Nr into populated systems includes synthetic fertilizer application to urban turfgrass and recreational areas (~1.1 Tg), and atmospheric deposition 0.2 Tg.

Much of the annual Nr input into these terrestrial systems passes through, and is transferred within, terrestrial systems or atmosphere via NH₃, NO_x or N₂O,

or aquatic environmental systems via NO₃⁻ and organic N leaching and runoff or NH_x and NO_y deposition.

The largest single reservoir of total N in the terrestrial environmental system is soil organic matter (SOM). Approximately 52,000 Tg C and 4,300 Tg N are contained in the upper 100 cm of soil in the 48 contiguous states (N is estimated from assumed C/N ratio of 12) (Lal et al., 1998). For comparison, the total above ground biomass of U.S. forests of these states contains ~ 15,300 Tg of C and ~ 59 Tg N (estimated using a C/N ratio of 261), and 15,500 Tg of soil organic matter carbon (SOM-C), and 1,290 Tg total N (estimated using a C/N ratio of 12) (U.S. EPA, 2007e). Most of this soil organic matter nitrogen (SOM-N) is bound within complex organic molecules that remain in the soil for tens to thousands of years. A small fraction of this SOM is mineralized, converted to carbon dioxide and Nr annually. The total N contained within above and below ground compartments isn't really of concern. What is of interest in addressing issues of Nr, is the change in N stored within the compartments of terrestrial systems. The pertinent question is whether N is being retained or released from long-term storage. The Committee evaluated estimates of annual change of N storage within important components of terrestrial systems. The Committee used Carbon stock information in Chapter 7 of the EPA Inventory of U.S. Greenhouse Gas Emissions and Sinks 1990-2005 (U.S. EPA, 2007e) to estimate N storage in U.S. terrestrial systems. Nitrogen stock change was determined by simply assigning a molar C/N ratio of 12 for soils and 261 for trees and making the appropriate conversions from C to N.

Agricultural systems

Croplands within the contiguous 48 states occupy ~368 million acres (149 million ha) (19%) of the 785 million ha of total land area. In 2002, 126 million ha of this cropland were cultivated (USDA NRCS, 2007). Croplands are generally found on well drained mineral soils (organic C content 1-6% in the top 30 cm). Small areas of drained organic soils are cultivated (organic C content of 10-20%) in mainly Florida, Michigan and Minnesota (EPA, 2007e). Organic soils lost ~0.69 Tg of Nr in 2002 while mineral soils accumulated ~1.5 Tg of Nr (Table 10). Much of the accumulation of SOM-C was due to the use of conservation tillage and high yielding crop varieties (U.S. EPA, 2007e). Losses of Nr from organic soils are due to mineralization of SOM and release of Nr input. In cultivated soils annual input of new Nr is approximately 9.7 Tg from fertilizer N, 1.1 Tg from livestock manure (recycled N), ~7.7 Tg from biological N fixation, and 1.2 Tg from atmospheric deposition. Assuming that loss of fertilizer N from the small area of organic soils is a minor fraction of the total, ~17% of N input from synthetic fertilizer, ~12% of total N input, is stored in cropland mineral soils annually.

According to the U.S. EPA National Greenhouse Inventory (U.S. EPA, 2007g) the net increase in soil C stocks over the period from 1990 through 2005 was largely due to an increase in annual cropland enrolled in the Conservation Reserve Program, intensification of crop production by limiting the use of bare-summer fallow in semi-arid regions, increased hay production, and adoption of conservation tillage (i.e., reduced- and no-till practices). It is clear that conversion of marginal crop land to CRP stores C and reduces erosion and nitrate leaching. Likewise, use of soil conservation tillage, as opposed to conventional tillage with a plow or disk, reduces erosion. However, the impact of conservation tillage on soil C storage and Nr losses due to leaching or denitrification are much less certain. For example, although the EPA estimates shown in Table 10 assume that no-till crop production results in net carbon sequestration, recent publications indicate that no-till cropping practices do not result in net carbon sequestration (Verma et al., 2005; Baker et al. 2007; Blanco-Canqui and Lal, 2008). Therefore, the estimates of soil C and N storage in mineral soils in Table 10 that were derived from U.S. EPA, (2007b) need to be reconsidered. These new studies and that of David et al. (2009) suggest that organic C conservation by reduced tillage practices has been overestimated because soil sampling and analysis has been confined to the top 30 cm of soil when the top meter of soil needs to be considered. Baker et al. (2007) and Verma et al. (2005) also show that long-term, continuous gas exchange measurements have not detected C gain due to no-till practices. They concluded that although there are other good reasons to use no-till, evidence that it promotes C sequestration is not compelling. These findings highlight the need for appropriate assessment of ecosystem N storage

in order to confirm or disprove this Committee's conclusion that only a small part of annual Nr input is stored in agricultural lands, forests, and grasslands.

Populated systems – urban lands

Populated or “developed land” (developed land is the terminology used by the U.S. Department of Agriculture's Natural Resources Conservation Service [NRCS]) occupied ~106 million acres (42.9 million ha) of the U.S. land area in 2002. This equates to approximately 5.5% of the U.S. land area (USDA NRCS, 2007). The EPA Inventory of U.S. Greenhouse Gas Emissions and Sinks (U.S. Environmental Protection Agency, 2007e) indicates that urban areas cover over 4.4% of the land area with tree canopy covering 27.1% of the urban area. The tree-covered area constitutes approximately 3% of total tree cover in the continental U.S. If the NRCS value of 42.9 million ha is used, then trees cover ~27.9 million acres (11.3 million ha) of urban land in the contiguous 48 states. Another ~35.1 million acres (14.2 million ha) of land is covered by turf grass in parks, golf courses, and lawns. In both urban forests and turf grass, Nr storage is dependent upon the age of the trees or turf. In young (pre-steady state) systems N is being accumulated, while at steady state no net change occurs. Some areas may be degrading and actually losing biomass and returning N to the environment. EPA does not estimate carbon changes in turf grass, but does estimate changes in carbon storage in urban forests (U.S. EPA, 2007e). Urban trees sequestered an estimated net 22 Tg of carbon and 0.12 Tg of N in 2002 (assuming that urban trees are mainly hardwood having a C/N ratio of 186) (U.S. EPA, 2007e). Annual fertilizer N input into the urban landscape is approximately 10% of total fertilizer N consumption in the U.S. (U.S. EPA, 2007e), or ~1 Tg of N in 2002. Another 0.2-1.0 Tg N is deposited from the atmosphere. In some areas deposition can be disproportionately high due to locally high NO_y concentrations. Storage of ~0.12 Tg N in urban forests constituted approximately 3% of Nr input annually.

Vegetated systems – forests and grasslands

Forests cover approximately 164 million ha, ~21% of the land area of the contiguous 48 states (USDA NRCS, 2007). The forest carbon stocks analysis by EPA (U.S. EPA, 2007e) is based on state surveys that are conducted every 1 to 10 years. Annual averages are applied to years between surveys. To determine changes in C related to the rate of tree growth, Birdsey (1992) estimated that there were 52,500 Tg of C above and below ground in U.S. forests; soil contained 59% of total C, 9% was in litter, and 5% in tree roots. The EPA estimate for 2002 was 43,600 Tg of C. To estimate N storage based on EPA data (U.S. EPA, 2007e), the Committee has assumed that forests are 85% softwood and 15% hardwood with an average C/N ratio of 261. These estimates indicate that forests and forest products stored ~0.43 Tg of N in 2002 (Table 10).

Grasslands, including rangelands and pasturelands, occupy approximately 213 million ha (27.1%) of the contiguous 48-state land area. The NRCS divides these grasslands into pastureland (48.2 million ha) and rangeland (164 million ha). Pastureland is managed (it may be fertilized and mown) and rangeland is managed only to the extent that livestock grazing intensity is regulated on the land used for such grazing. Changes in the N status of grasslands are dependent upon changes in soil organic matter as the above ground biomass produced annually is either consumed by livestock or decomposed in the field. Soil organic C stocks were estimated using

the Century biogeochemical model and data used were based upon the NRCS/National Resources Inventory (NRI) survey (U.S. EPA, 2007e). Changes in soil N content were estimated using a C/N ratio of 12. Nitrogen input into rangelands is generally only from atmospheric deposition, which contributes 1.9 Tg N each year to range production. Rangeland tends to be in relatively remote areas where atmospheric N_r deposition is low.

Collectively, forests and grasslands stored ~0.74 Tg of N in 2002. Much of the soil N storage in grasslands is a result of conversion of croplands to grasslands, mainly due to the conservation reserve program. Forest soils

Table 10: Net annual change in continental U.S. croplands soil N and C, forest C and N, and grasslands C and N in 2002

	C	N
Cropland		
Cropland remaining cropland		
Mineral soil	17	1.4*
Organic soil	-8.3	-0.69
Land converted to cropland	0.8	0.067
Total	9.5	0.80
Forests		
Forests and harvested wood products		
Above ground biomass	85	0.32
Below ground biomass	16	0.063
Dead wood	9.1	0.035
Litter	7.2	0.028
Soil organic matter	-2.8	-0.23
Harvested Wood	59	0.22
Total	174	0.44
Grasslands		
Grasslands remaining grasslands		
Mineral soil	-0.8	-0.067
Organic soil	-1.3	-0.11
Lands Converted to Grasslands	5.8	0.48
Total	3.7	0.31
US Total C & N Storage in 2002	187	1.5

Measurements in Tg. Negative sign indicates a decrease in storage; positive number indicates increase in storage. Soil C/N ratio = 12; wood C/N = 261. C storage numbers were obtained from U.S. EPA, 2007e.* See previous discussion of soil organic matter accumulation in croplands.

appear to be losing N while overall N storage in forests is from accumulation in above-ground biomass and that that remains in forest products that are stored for long periods.

Summary of estimates of Nr stored in terrestrial systems in 2002

Although our estimate of N storage in terrestrial systems is highly uncertain, annual N storage is likely relatively small. Approximately 1.5 Tg N was stored in terrestrial systems of the contiguous 48 states in 2001 (Table 10). Soils were the largest reservoir with croplands (0.8) and grasslands (0.3) sequestering most of the N. Annual storage in agricultural, grassland and forest soil and in forest biomass is approximately 6 to 10% of annual Nr input. Estimated total Nr input from synthetic fertilizer, biological N fixation, and atmospheric deposition into terrestrial systems within the contiguous 48 states in 2002 was ~32 Tg. All of the input and outflow numbers are highly uncertain, but N loss through denitrification appears to be the major loss mechanism. Storage in soils and trees appears to account for only a small portion of the annual N input while apparent loss through denitrification dominates the budget. This is discussed in Section 2.3.3 of this report and in a recent global Nr review by Schlesinger (2009). Some small fraction is re-volatilized and exported from the continent.

2.3.3. Areas of Uncertainty in Nr Transfer and Transformation

In considering Nr transfers and transformations in and between the environmental systems of the nitrogen cascade, the Committee has encountered a number of areas where quantities or flows of Nr are highly uncertain. All of these areas need attention from EPA in conjunction with other federal and state agencies and universities. Although most of the following points have been highlighted in various findings and recommendations within other chapters of this report, we feel the need to highlight the following areas:

- Total denitrification in animal feeding operations, in soils, and in aquatic systems needs to be quantified along with all gaseous products produced and released to the atmosphere during nitrification/denitrification. These gases include NO_x, N₂O and N₂.
- The amount of Nr transferred to each environmental system as dry deposition needs to be quantified and monitored.
- The fraction of NO_y in the form of organo-nitrates and other organic nitrogen species is poorly quantified, but may play a major role in air quality and Nr cycling.
- Rates and amounts of ammonia emissions from fertilized soils and animal feeding operations need to be quantified and the fate of this ammonia determined.
- The annual change in N storage in soils (agricultural, forest, grassland and urban areas) needs to be quantified in conjunction with the change in carbon.

These areas of high uncertainty are highlighted because very little information exists in some of the areas. In other areas, such as denitrification and the relative release of N₂O from soils and aquatic systems, the sparse data are highly variable and this makes developing meaningful guidelines for control difficult. An analysis of Nr input and fate in 16 watersheds in the U.S. is provided as an example to illustrate how the inputs and fate of Nr can be evaluated for a large watershed.

Input and fate of Nr in 16 watersheds in the northeast United States

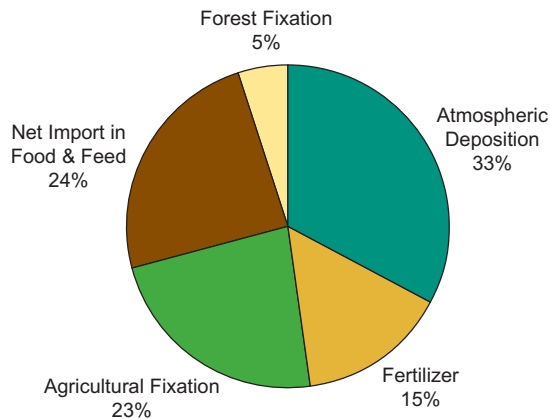
There are no comprehensive data available to assess the transfer and transformations in and between the atmosphere, terrestrial systems (agriculture, populated, and vegetated systems) and aquatic systems nationally. Determining a national N budget is a priority research area. As there are no national data available, an example analysis of Nr input and fate in 16 watersheds in the northeast U.S. (for which data are available) is used to show an evaluation of the inputs and fate of Nr for a large watershed (Van Breemen et al., 2002).

The watersheds in this study encompass a range of climatic variability, Maine to Virginia. The watersheds are a major drainage to the coast of the North Atlantic Ocean. Using data from the early 1990s, Boyer et al. (2002) reported the quantification of N inputs to each watershed from atmospheric deposition, N fertilizers, biological N fixation, and import of N in agricultural products (food and feed). They compared inputs with N losses from the system in riverine export. As a part of the same study, Van Breemen et al. (2002) analyzed the fate of N inputs to these watersheds and developed budgets for each watershed. The total area of the watersheds was 32,666 square km with land use categories of forest (72%), agricultural (19%), urban (3%), wetlands (5%), and 1% other uses. The Nr input into the watersheds (using weighted averages for all 16 watersheds) was 3,420 kg per square km per year. Figure 14 shows the Nr sources and the estimated fate of this Nr as a per cent of the weighted average Nr input.

Van Breemen et al. (2002) indicate that Nr inputs and storages and losses were well correlated ($R^2 = 0.98$). Denitrification in landscape soils is the most uncertain estimate because rates are calculated by difference between total inputs and outputs so they accumulate errors from all estimates. The denitrification loss term may also reflect the change in N storage in groundwater. The net storage of N in the soil (18% of total storage and losses) indicates that there is a non-steady state condition in the soil. Increasing storage of Nr on land implies that drainage and denitrification exports of Nr are likely to increase when a new steady-state condition is reached.

These data suggest that Nr research needs to focus on understanding the “denitrification” loss term in this analysis. The losses occur in the terrestrial landscape, before Nr

Nr Sources



Nr Storage & Loss

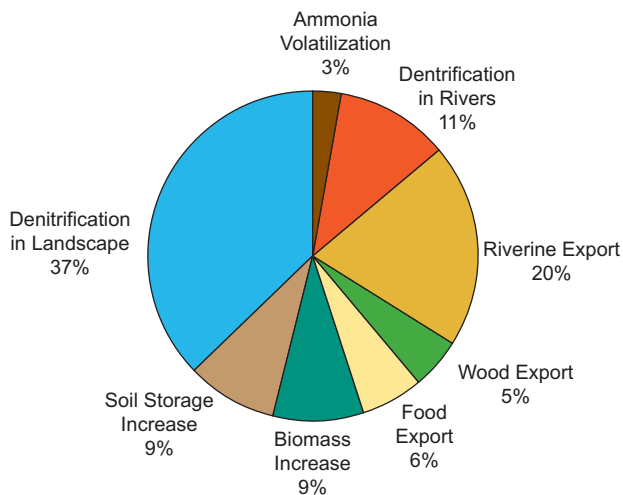


Figure 14: Nr input and loss from 16 watersheds in the northeast United States

Source: Van Breeman et al., 2002 (Figure 7, p. 289). Reprinted with permission; copyright 2002, Springer Science+Business Media B.V.

enters the river. Where do these losses occur, within the agricultural field, in drains and ditches near the agricultural field, in riparian areas, or wetlands? Understanding this term may help in the management of Nr in watersheds to decrease nitrate movement into aquatic systems as well as to limit N_2O emissions to the atmosphere.

The Van Breeman et al. (2002) study also estimated that approximately 30% of N input was exported to the rivers and about two thirds (20% of total N input) of this N was exported to coastal waters by rivers. The remaining one-third (11% of total N input) was considered to have been denitrified in the rivers. These examples also demonstrate that Nr in the atmosphere, terrestrial systems and aquatic systems are not separate and must be considered collectively. Atmospheric deposition is a variable but important input into aquatic systems that contributes to Nr enrichment problems. Aquatic and terrestrial systems process this Nr and return other Nr gases (NH_3 , NO_x and N_2O to the atmosphere). Nr from terrestrial systems impacts both the atmosphere and aquatic systems through emission of NH_3 , NO_x , N_2O and leaching and runoff of NO_3^- .

Finding 10: Denitrification of Nr in terrestrial and aquatic systems is one of the most uncertain parts of the nitrogen cycle. Denitrification is generally considered to be a dominant N loss pathway in both terrestrial and aquatic systems, but it is poorly quantified.

Recommendation 10: EPA, USDA, DOE, and universities should work together to ensure that denitrification in soils and aquatic systems is properly quantified, by funding appropriate research.

